National Park Service
U.S. Department of the Interior

Natural Resource Stewardship and Science

Nearshore Marine Vital Signs Monitoring in the Southwest Alaska Network of National Parks

2012

Nearshore Marine Vital Signs Monitoring in the Southwest Alaska Network of National Parks

2012

Heather A. Coletti

National Park Service
240 W 5th Avenue
Anchorage, AK 99501

Tom Dean[1], Kimberly Kloecker[2], Brenda Ballachey[2]

[1]Coastal Resources Associates Inc.
5190 El Arbol Drive
Carlsbad, CA 92008

[2]US Geological Survey
Alaska Science Center
4210 University Drive
Anchorage, AK 99508

The National Park Service, Natural Resource Stewardship and Science office in Fort Collins, Colorado, publishes a range of reports that address natural resource topics. These reports are of interest and applicability to a broad audience in the National Park Service and others in natural resource management, including scientists, conservation and environmental constituencies, and the public.

The Natural Resource Technical Report Series is used to disseminate results of scientific studies in the physical, biological, and social sciences for both the advancement of science and the achievement of the National Park Service mission. The series provides contributors with a forum for displaying comprehensive data that are often deleted from journals because of page limitations.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible, technically accurate, appropriately written for the intended audience, and designed and published in a professional manner.

This report received informal peer review by subject-matter experts who were not directly involved in the collection, analysis, or reporting of the data.

Views, statements, findings, conclusions, recommendations, and data in this report do not necessarily reflect views and policies of the National Park Service, U.S. Department of the Interior. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the U.S. Government.

This report is available in digital format from (http://science.nature.nps.gov/im/units/swan/) and the Natural Resource Publications Management website (http://www.nature.nps.gov/publications/nrpm/). To receive this report in a format optimized for screen readers, please email irma@nps.gov.

Please cite this publication as:

Coletti, H. A., T. A. Dean, K. A. Kloecker and B. E. Ballachey. 2014. Nearshore marine vital signs monitoring in the Southwest Alaska Network of National Parks: 2012. Natural Resource Technical Report NPS/SWAN/NRTR—2014/843. National Park Service, Fort Collins, Colorado.

Contents

Contents (continued)

Figures

Figures (continued)

Figures (continued)

Tables

Abstract

In 2012, we successfully completed another year of field sampling for the Southwest Alaska Network's (SWAN) Nearshore Vital Signs monitoring program and the Gulf Watch Alaska Program in accordance with standard operating procedures set forth for the six vital signs: marine intertidal invertebrates, kelp and seagrass, marine water chemistry and quality, marine birds, black oystercatcher, and sea otter. All analyses in this report are descriptive and no statistical tests were used in any comparisons across regions or time for this entire report.

Summer sampling in 2012 represented the sixth year of data collection at Katmai National Park and Preserve (KATM) and Kenai Fjords National Park (KEFJ) and the fourth year of sampling in western Prince William Sound (WPWS). Sampling did not occur in KATM in 2011 or in WPWS in 2008 or 2009. We anticipate continued annual sampling in the SWAN parks as well as WPWS for the vital signs: intertidal invertebrates, kelps and seagrasses, water chemistry and quality, marine bird surveys (only KATM and KEFJ), black oystercatcher diet and productivity, and sea otter diet. Data from WPWS are presented here in anticipation that all three areas (KATM, KEFJ and WPWS) will continue to be sampled and analyzed together in order to provide a larger spatial context to the analyses. In addition to WPWS, monitoring sites area being added in eastern and northern PWS and will be included in future analyses.

No modifications were made to the rocky intertidal sampling protocol from previous years and the protocol and SOPs have been finalized. Hobo water temperature sensors are currently deployed at five rocky intertidal sites in each area (KATM, KEFJ and WPWS). In addition, salinity loggers are co-located at rocky intertidal sites at KATM, KEFJ and WPWS. We implemented a fifth year of mussel bed and eelgrass bed sampling and a final SOP for sampling mussel beds is near completion and will be sent out for peer review in 2013-2014. Modifications for eelgrass bed monitoring are being made and a new draft SOP will be sent for review.

Marine bird surveys in KEFJ and KATM continued with little modification in 2012. For marine bird surveys, we recommend that the survey effort continue until further analysis can be completed. The existing SOP for marine bird surveys is final.

Black oystercatcher abundance, nest density, productivity and diet data should continue to be collected with little revision. Sampling at the current intensity should allow us to detect trends in changes of nest density, productivity and diet (especially prey size) of the black oystercatcher. The SOP for black oystercatcher monitoring is also final.

A sea otter aerial survey was completed in KATM during August of 2012. This was the second aerial survey completed along the KATM coastline. An aerial survey was previously flown in 2008. Survey methodology followed Bodkin and Udevitz (1999) and accounts for imperfect detection. The survey took three days to complete. The estimated sea otter abundance for KATM is 8,644 individuals, with an overall density of 5.95 otters/km^2. The 2008 abundance estimate was 7,095 individuals with an estimated density of 4.89 otters/km^2.

Sea otter foraging data was collected in KATM and KEFJ in 2012. Clams (multiple species) and mussels (*Mytilus trossulus*) dominated sea otter diets across all years of data collection (2007-2012), together comprising over 80% of the diet. Annually there has been little observed change in the predominant prey category at any area, although sea otters observed at KEFJ in 2012 consumed more clams and fewer mussels. A sea otter forage database has been completed. Database completion will

ease data entry both in the field and office as well as optimize data analysis. Carcass collection continues in all areas, although to date we have not recovered enough carcasses from KEFJ to employ age-specific mortality analyses.

In the spring of 2014 we will finalize data entry and data management procedures for the sea otter foraging, mussel and soft sediment SOPs. We will continue to sample nearshore vital signs at KATM, KEFJ and WPWS in 2014.

In 2014, the protocol narrative for the cooperative SWAN and Gulf Watch Alaska monitoring program was updated through an external peer review process. http://science.nature.nps.gov/im/units/swan/assets/docs/reports/protocols/nearshore/DeanT_2014 _SWAN_NearshoreProtocolNarrative.pdf

Acknowledgments

Intertidal Invertebrates and Algae Sampling

Introduction

Intertidal invertebrate and algal communities provide an important source of production; are an important conduit of energy, nutrients, and pollutants between terrestrial and marine environments; provide resources for subsistence, sport, and commercial harvests; and are important for recreational activities such as wildlife viewing and fishing. The intertidal is particularly susceptible to human disturbance including oil spills; trampling by recreational visitors; harvesting activities; pollutants from terrestrial, airborne and marine sources; and shoreline development. Changes in the structure of the intertidal community serve as valuable indicators of disturbance, both natural (e.g. Dayton 1971, Sousa 1979) and human induced (Barry et al. 1995, Lewis 1996, Keough and Quinn 1998, Jamieson et al. 1998, Shiel and Taylor 1999, Sagarin et al. 1999, Peterson 2001, and Peterson et al. 2003).

Intertidal invertebrates and algae (including intertidal kelps) were sampled annually at KATM beginning in 2006; however no sampling occurred in KATM in 2011. Sampling began in KEFJ in 2008. WPWS sampling began in 2007 and then again from 2010-2011. Sampling of intertidal invertebrates and algae at these sites is designed to detect changes in these communities over time as part of the NPS SWAN Vital Signs Program and Gulf Watch Alaska Monitoring Program. The specific objectives of this sampling on rocky shores are to assess changes in: 1) the relative abundance of algae, sessile invertebrates, and motile invertebrates in the intertidal zone, 2) the diversity of algae and invertebrates, 3) the size distribution of limpets (*Lottia persona*) and mussels (*Mytilus trossulus*), 4) the concentration of contaminants in mussel tissue, and 5) temperature (either sea or air depending on tidal stage). In this section, we present results of sampling conducted in 2006-2011. The metrics to be examined are: 1) abundance estimates for dominant taxa of sessile invertebrates and algae, and the size distribution of the limpet *Lottia persona*.

Methods

Sampling was conducted at five sites in sheltered rocky habitats within KATM, KEFJ and WPWS. Descriptions of the study sites and methods used to sample intertidal algae and invertebrates are available in Dean and Bodkin (2011b). Sites were chosen using a generalized random tessellation stratification (GRTS) procedure (Stevens and Olsen, 2004) that provided a spatially balanced yet random selection of sites. The following is a general description of the methods employed. Sampling of abundance and species composition for algae and invertebrates was conducted along two 50 m linear transects at each site along the 0.5m and 1.5m tidal elevations. The percent cover of algae and sessile invertebrates was estimated within 12 evenly spaced ¼ m^2 quadrats placed along transects that ran parallel to the shoreline and originated at permanent markers, respectively. Quadrats were placed at random start points and at equally spaced intervals thereafter. In addition, a minimum of 120 individual limpets (*Lottia persona*) were measured at each site for estimation of size distributions.

The analyses presented here focus on estimates of abundance of dominant taxa at each tidal elevation, and on size distributions of limpets. Means and 95% confidence intervals are reported for each park in each year.

Results

Mean percent cover (and 95% confidence intervals) are reported for each site at KATM, KEFJ and WPWS in Figures 1 through 9. Relative abundance varied by region and tidal elevation, but *Fucus distichus. evanescens,* barnacles, and *Alaria marginata* were generally the most abundant. *Alaria marginata* does not occur at the higher tidal elevation (1.5 m) and does not generally occur in WPWS. *Alaria marginata* occurs along coastlines with more exposure than those found in WPWS. Notable differences between regions were observed at the lower (0.5 m MLLW – mean low low water) tidal elevation, with a greater percent cover of *Fucus* at KEFJ. The only notable trend over time was an increase in cover by *Fucus* at the 1.5 m tidal elevations at KEFJ between 2008 and 2011. No differences between regions were noted for the mean size of the limpet *Lottia persona* (Figure 10).
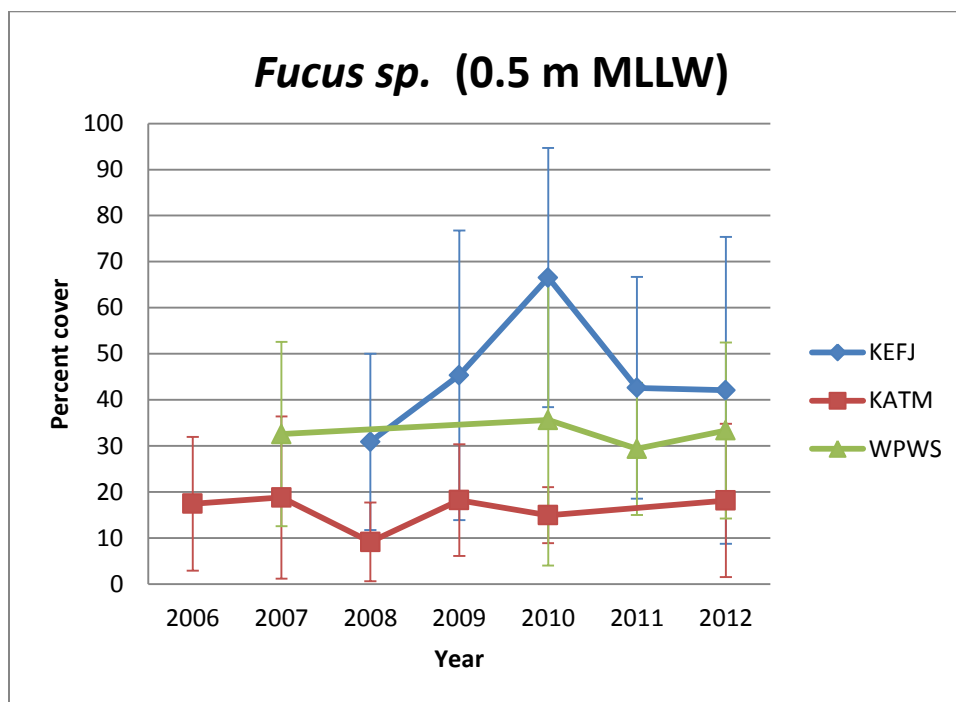
Figure 1. Percent cover of *Fucus* at the 0.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
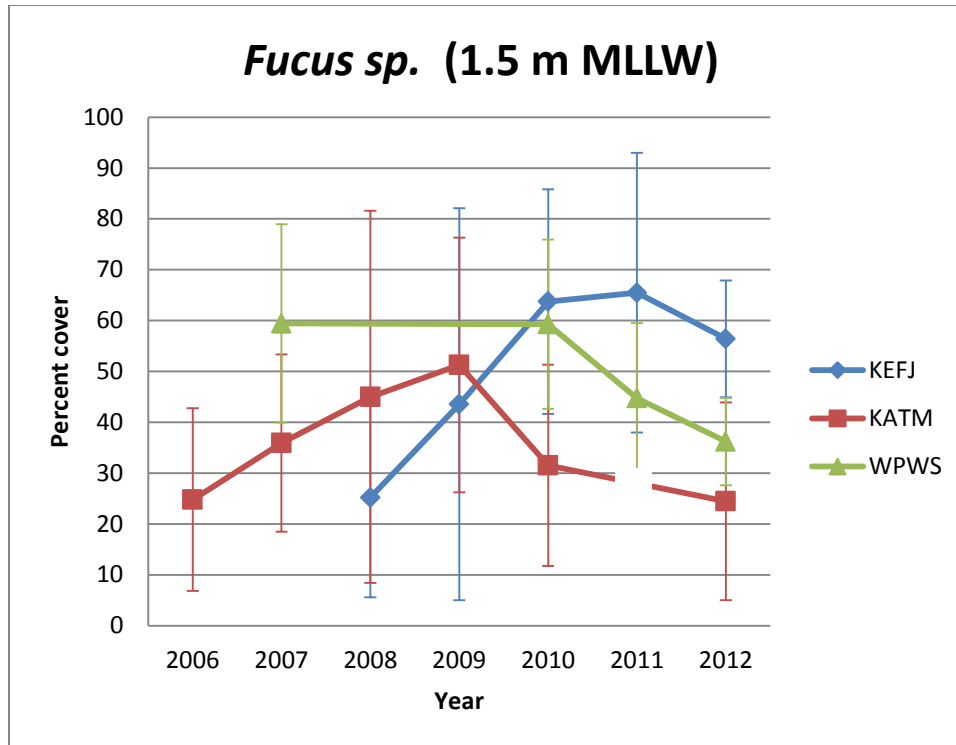
Figure 2. Percent cover of *Fucus* at the 1.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
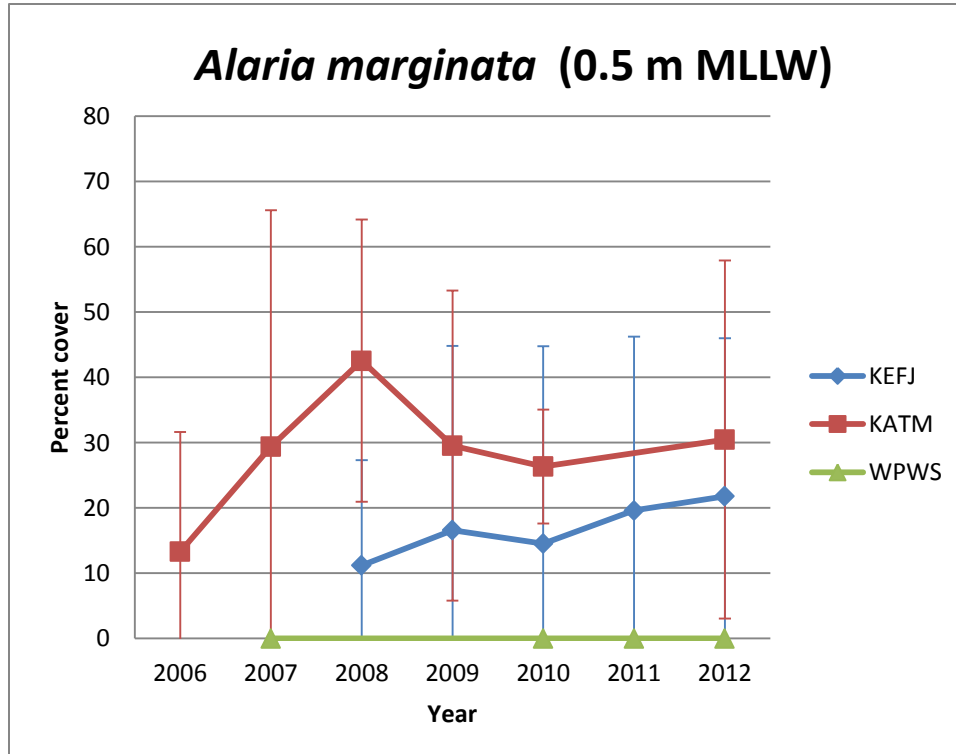
Figure 3. Percent cover of *Alaria* at the 0.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
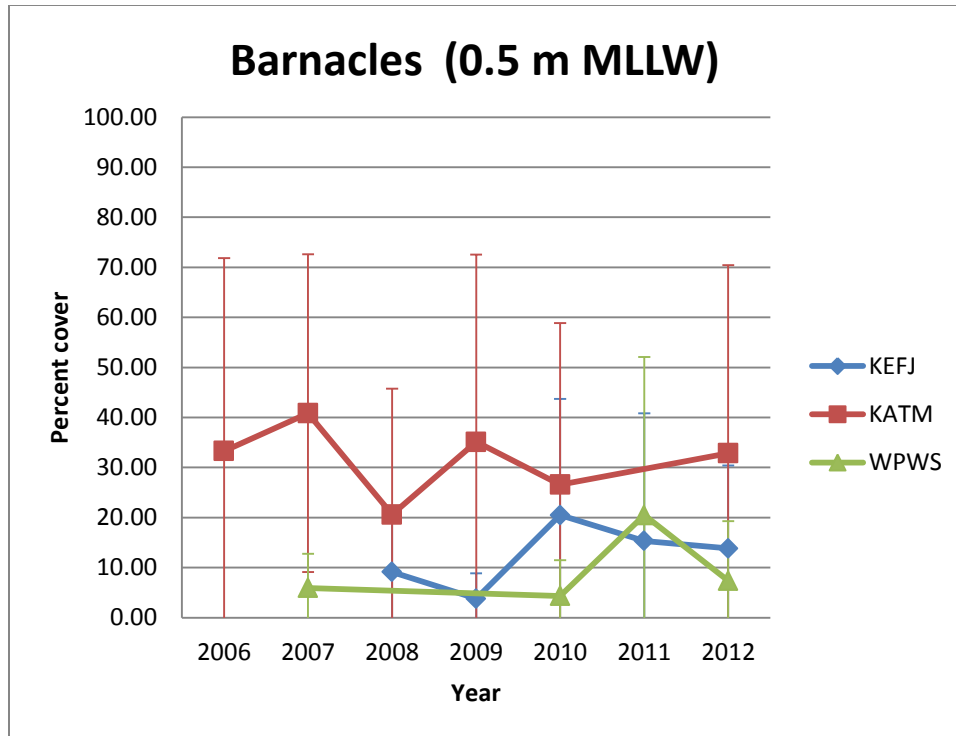
Figure 4. Percent cover of barnacles at the 0.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
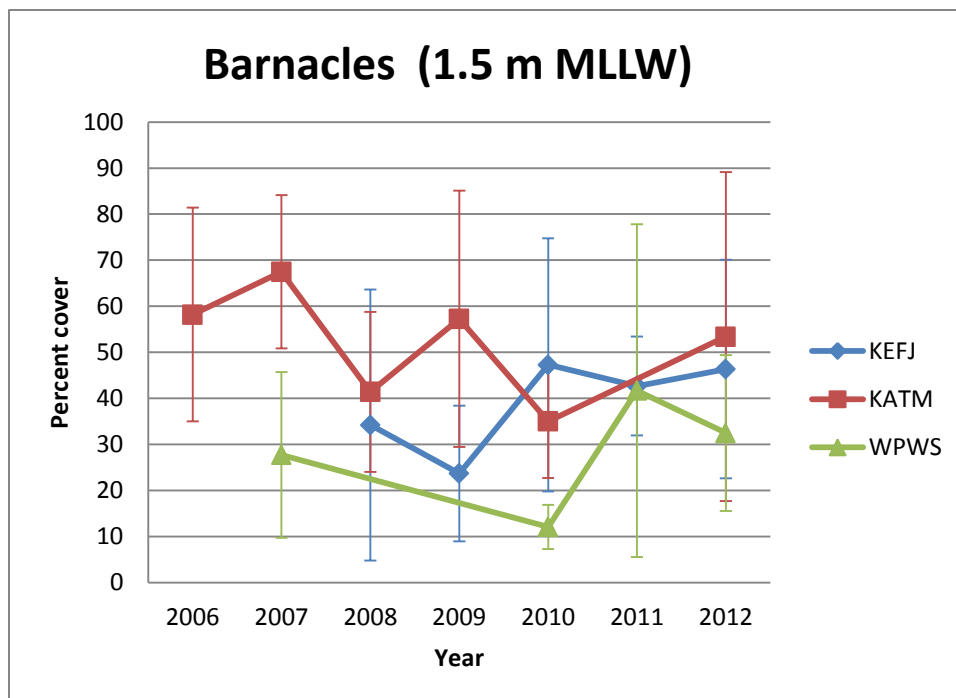
Figure 5. Percent cover of barnacles at the 1.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
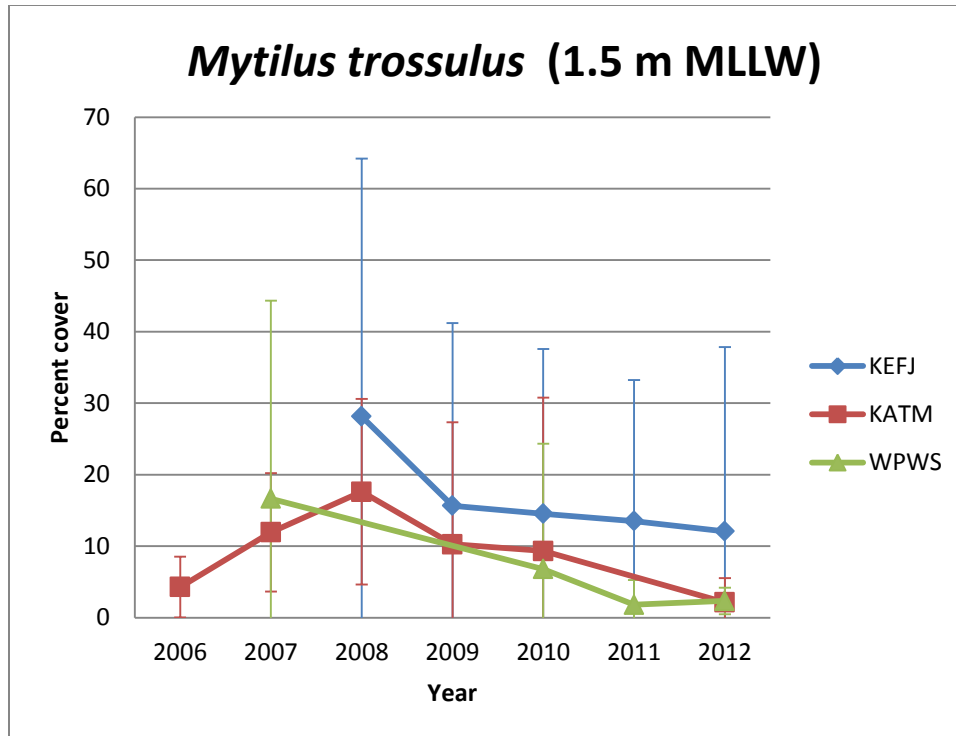
Figure 6. Percent cover of *Mytilus* at the 1.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
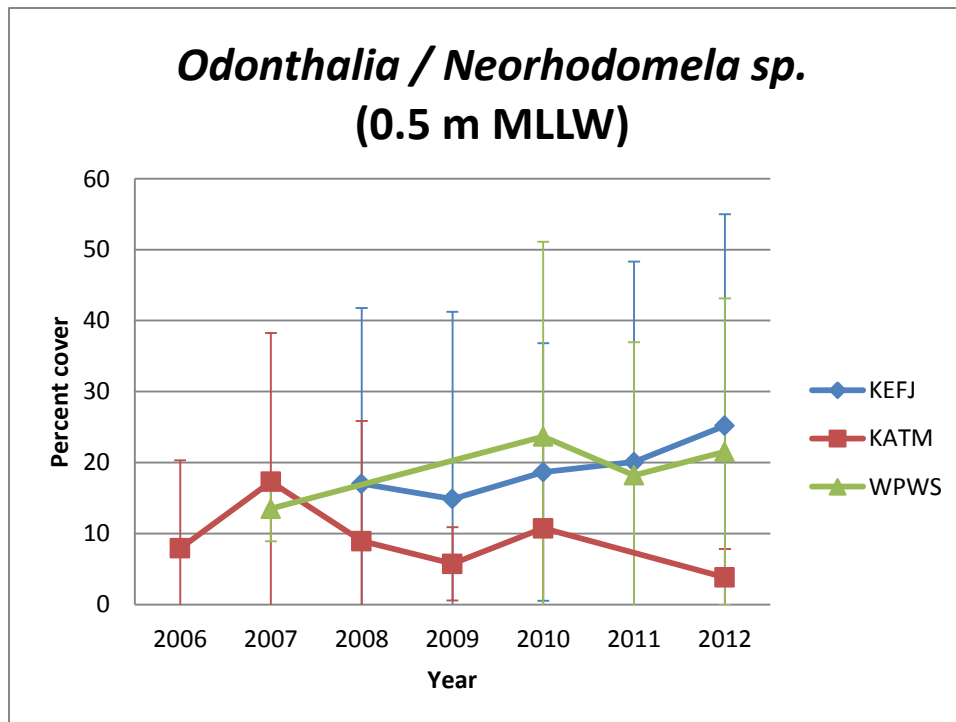
Figure 7. Percent cover of *Odonthalia / Neorhodomela* at the 0.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
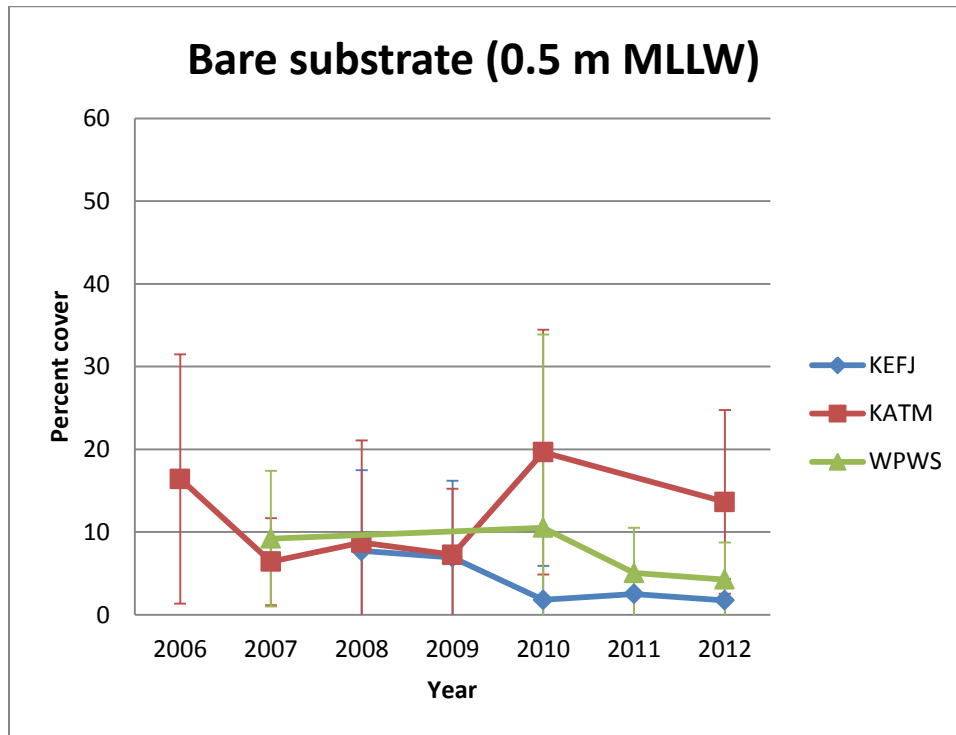
Figure 8. Percent cover of bare substrate at the 0.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
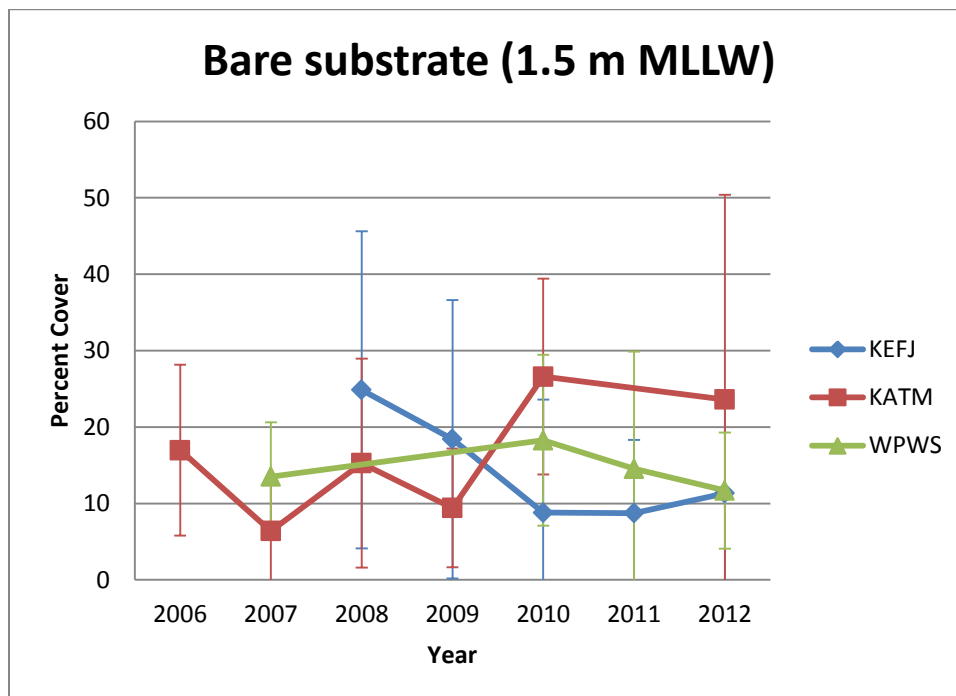
Figure 9. Percent cover of bare substrate at the 1.5 m MLLW in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
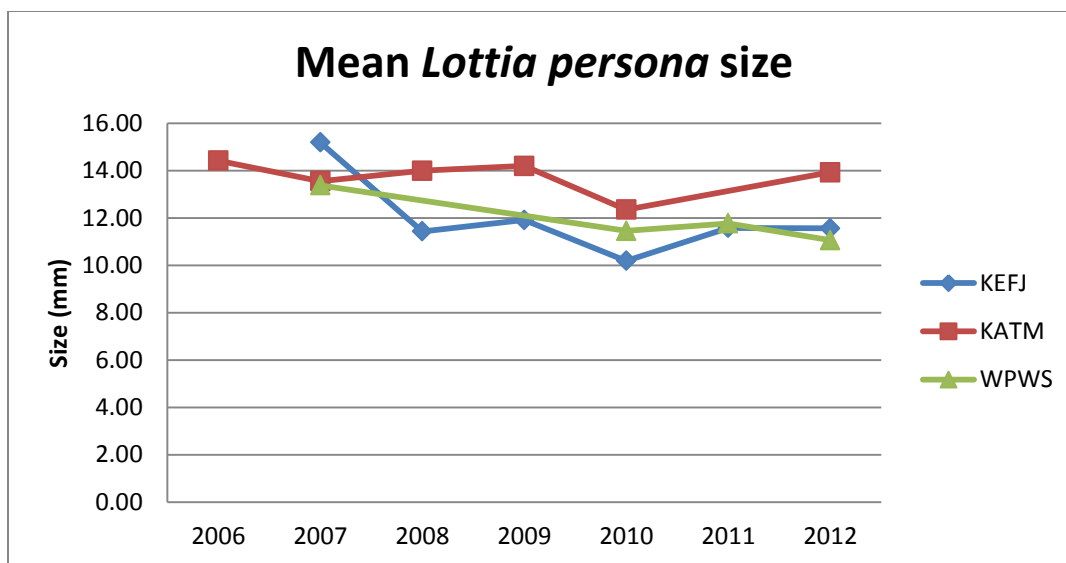
Figure 10. Mean size (mm) of *Lottia persona* in KATM, KEFJ and WPWS, 2006-2012. Error bars indicate 95% CI.

Discussion

Sampling provided estimates of the abundance of intertidal invertebrates and algae (including intertidal kelps) at sites within each region. We anticipate that the methods employed will detect ecologically meaningful levels of change in the future. Existing data will allow the program to conduct trend analysis for several metrics and build simulations to estimate the number of samples and sample frequency required to detect a specified trend or change with some level of confidence for selected metrics, specifically the rocky intertidal algae and invertebrate vital sign. The rocky intertidal invertebrate and algae vital sign has eight metrics that have several years of data to conduct simulations to determine the power to detect change. The levels of change or trend to be detected have already been specified by the investigators (Dean and Bodkin 2011a, Dean et al. 2014). The Vital Signs Monitoring Plan for SWAN (Bennett et al 2006) explicitly states the use of hierarchical models to estimate trends. The work proposed here is to assist the National Park Service in the modification of the protocol for its monitoring program.

Recommendations

Based on these results, we recommend continued estimation of percent cover by sessile invertebrates and algae using random point counts and continued estimation of sizes of limpets.

Mussel Bed Sampling

Introduction

Pacific blue mussels (*Mytilus trossulus*) are a dominant and ubiquitous invertebrate in the intertidal zone and are critically important prey for a variety of organisms including sea otters (*Enhydra lutris*), black oystercatchers (*Haematopus bachmani*), harlequin ducks (*Histrionicus histrionicus*), Barrow's goldeneyes (*Bucephala islandica*), and several species of sea stars (O'Clair and Rice 1985, O'Clair and O'Clair 1988, VanBlaricom 1988, Andres and Flaxa 1995, Esler et al. 2002, Bodkin et al. 2002). Mussels are widely distributed in many intertidal habitats, but also form relatively monotypic stands of larger individuals that are termed mussel beds. The goal of mussel bed sampling is to assess changes in the size of beds and in the size of mussels within those beds over time. These data are used primarily as an indicator of mussel abundance as prey for various predators (sea stars, sea ducks and sea otters). Specifically, the objectives are to estimate: 1) the density of mussels within a bed, 2) the density of large mussels (greater than or equal to 20 mm in length) within a bed, and 3) the size distribution of the large mussels within a bed (those generally consumed by black oystercatchers, sea ducks and sea otters). We define mussel beds as sites with relatively high densities of mussels. Specifically, mussel beds are defined as areas with greater than approximately 10% cover by mussels within contiguous 0.25 m^2 quadrats over areas of 100 m^2 or greater. Metrics used to evaluate change over time will include the area of individual mussel beds (in m^2), average density of large mussels, and the mean size of large mussels. In this report, we include results of sampling mussels at sites in KATM, KEFJ and WPWS.

Methods

Sampling sites are defined as 50 m of coastline with contiguous mussel beds. These sites were selected following intensive searches in 2008 for the presence of mussel beds adjacent to the randomly selected rocky intertidal sites (see intertidal invertebrates and algae section). The closest mussel bed to the randomly selected rocky intertidal site was selected for sampling.

A transect 50 m in length was established through the mid-point of the bed, relative to tidal elevation, and at the left end of the bed, as observed from the water. A permanent bolt was placed at this location and at approximately 5 m intervals along the 50 m length of the horizontal transect to establish the site for future sampling. Ten vertical transects were then established at systematic intervals based on a random start point (a different random start point is used each year) along the horizontal transect length, and the distance from the upper most margin of the bed to the lower margin (or the 0 m tidal elevation) was measured for each vertical transect.

Estimates of mussel density are made within quadrats that are randomly located along each vertical transect. Quadrat dimensions are dependent on the density of mussels $\geq$ 20 mm within 1 m of the predetermined random point along the vertical transect, and determined at the time of sampling. The quadrat size can range from .0025 m^2 to 1.00 m^2 (5 cm to 100 cm on a side) with the size dependent on obtaining a collection of at least 20 mussels $\geq$ 20 mm in length. This results in at least 200 mussels to estimate size distributions of large mussels at a site. All mussels $\geq$ 20 mm are collected from within the quadrat and later counted and measured, and densities of large mussels are

calculated. Densities of all mussels (of a size that is visually detectable, approximately 5 mm and greater) are estimated from a 2.54 cm radius (20.27 cm^2) core located at the same random number that defined the vertical quadrat, but on the opposite side of the tape from the origin of the large mussel quadrat.

Results

In 2012 we estimated the abundance and size of mussels at five mussel bed sites at five sites each in KEFJ and KATM for the fifth year in a row, and at WPWS for the third year in a row. Results for each area are represented here. In general, mussel density is greater in KEFJ than in KATM or WPWS for all mussels including the large mussels (Figures 11 and 12). Mean sizes of mussels ≥20mm are similar across all three areas (Figure 13). The proportion of large mussels appears to have decreased in KEFJ and KATM from 2010 to 2012, but remained relatively stable in WPWS (Figure 14).
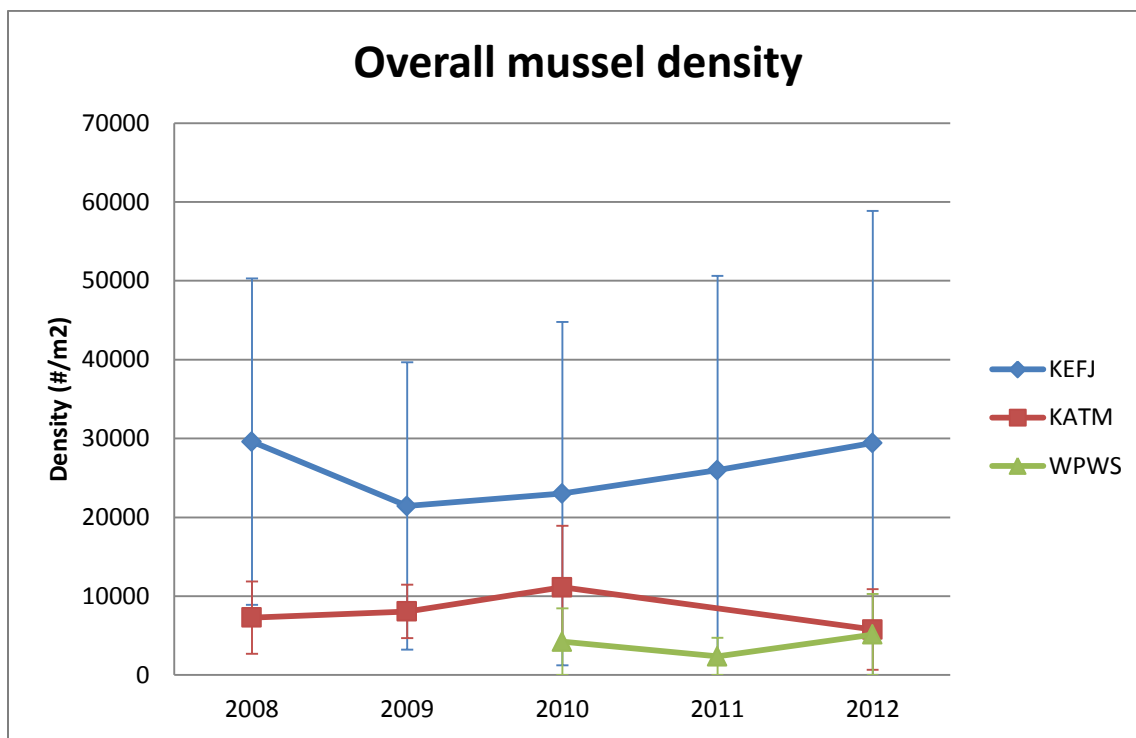
Figure 11. Overall mussel density (#/m^2) in KATM, KEFJ and WPWS, 2008-2012, based on core samples. Error bars indicate 90% CI.
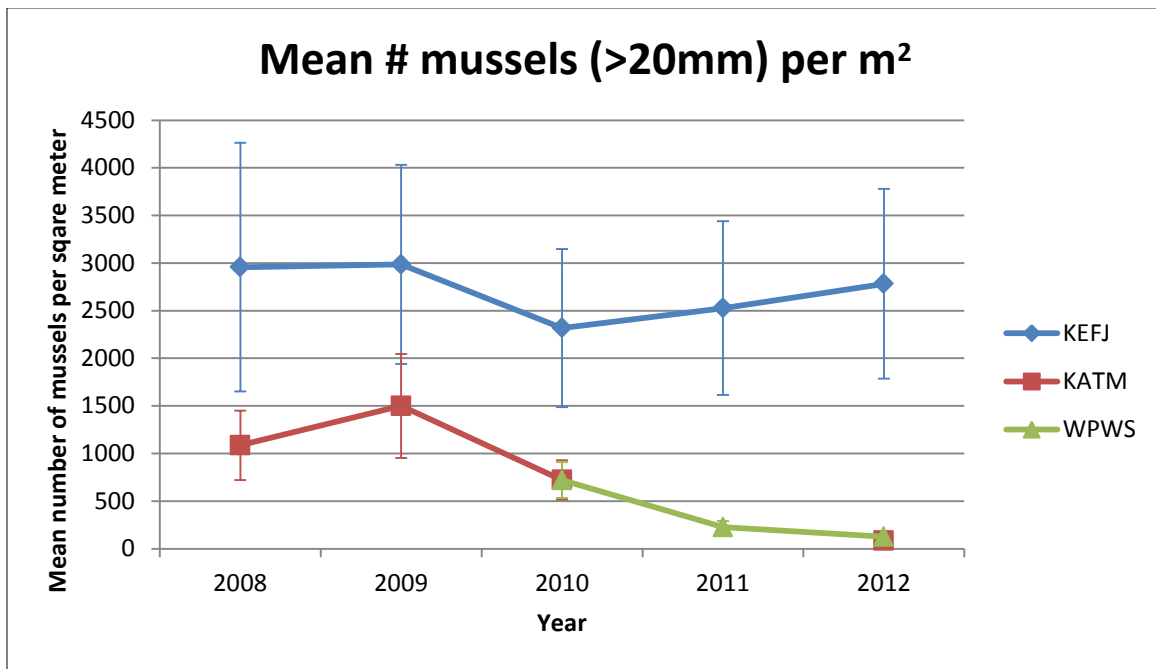
Figure 12. Density (#/m^2) of mussels ≥ 20 mm in KATM, KEFJ, and WPWS, 2008-2012. Error bars indicate 95% CI.
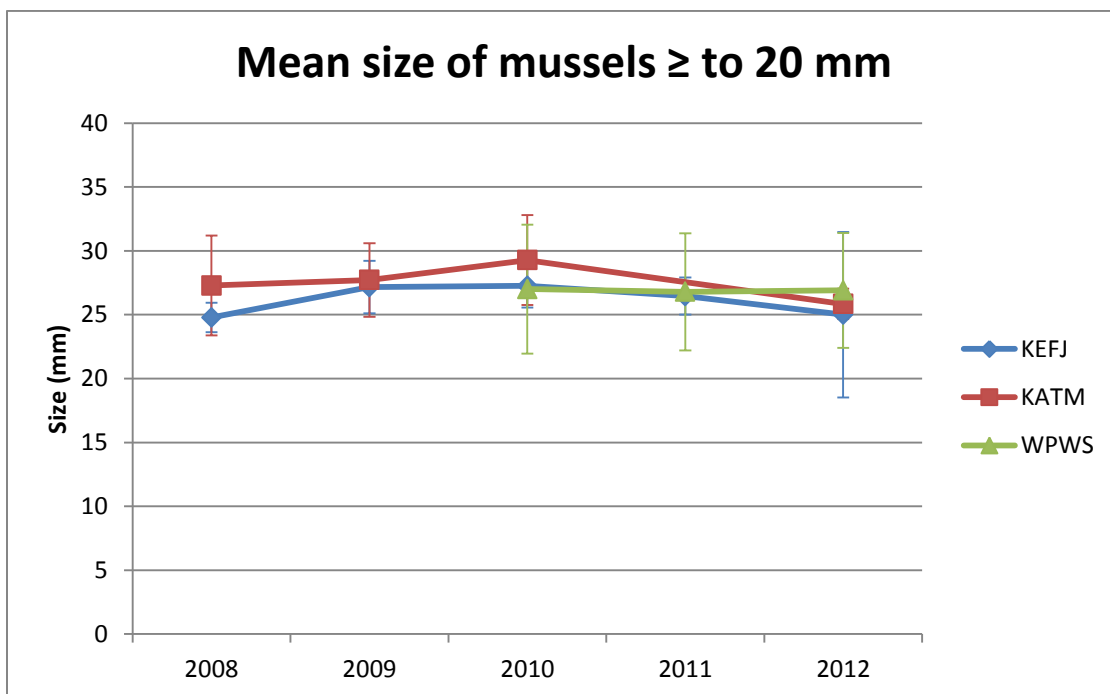
Figure 13. Mean mussel size in KATM, KEFJ, and WPWS, 2008-2012. Error bars indicate 95% CI.
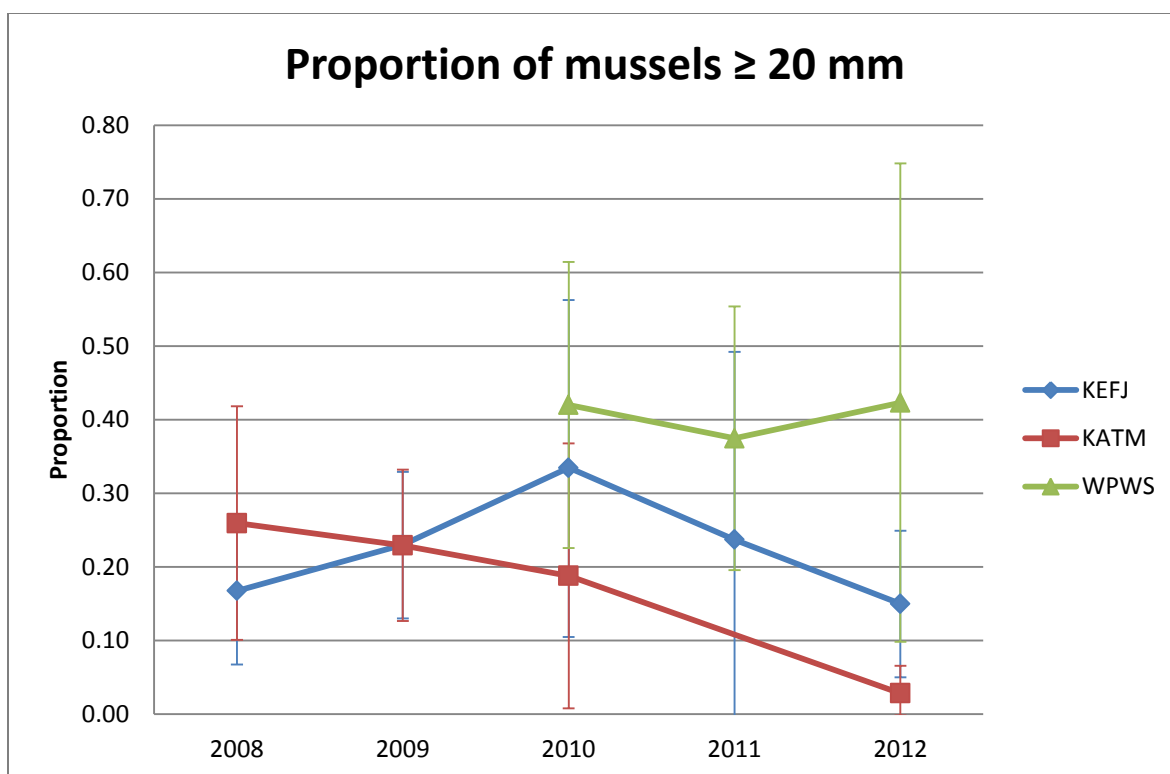
Figure 14. Proportion of mussels ≥ 20 mm in KATM, KEFJ, and WPWS, 2008-2012. Error bars indicate 90% CI.

Discussion

Using the methods briefly described above, we were able to estimate densities of mussels, the size distribution and density of large mussels (≥ 20 mm), and the proportion of large mussels. Mussel densities varied greatly between parks, both in terms of all mussels and large mussels. Mean sizes of large mussels were relatively uniform among all sites, indicated by the smaller error bars. The high uniformity in mean sizes and low variance among sites suggests common mechanisms structuring the sizes of mussels across all areas. While evaluating variance estimates of mussel densities and sizes for sensitivity to detect change will require additional years of data, the relatively low variation in mean sizes of large mussels across sites continues to suggest that mussel size may provide a statistically powerful metric to detect change over time.

Recommendations

Our fifth year of descriptive analysis indicates that sizes of mussels may provide a metric sensitive to change both among and within sites. We recommend the continuation of annual mussel bed sampling. Similar to the algae analysis discussed in the previous section, existing mussel bed data will allow the program to conduct trend analysis for several metrics and will be used in simulations to estimate number of samples and sample frequency required to detect a specified trend or change with some level of confidence for selected metrics, specifically the rocky intertidal algae and invertebrate vital sign. The levels of change or trend to be detected have already been specified by

the investigators (Dean and Bodkin 2011a, Dean et al. 2014). The Vital Signs Monitoring Plan for SWAN explicitly states the use of hierarchical models to estimate trends. The work proposed here is to assist the National Park Service in the modification of the protocol for its monitoring program. The SOP for mussel bed sampling will be finalized in 2014.

Eelgrass Bed Sampling

Introduction

Eelgrass (*Zostera marina*) is the dominant native seagrass in protected waters of the Gulf of Alaska and is broadly distributed in sheltered embayments, especially in habitats dominated by soft sediments where they often form "beds" or relatively monotypic stands that can cover much of the shallow (0 to 5 m depth) subtidal zone (McRoy 1968, 1970). Eelgrass is an important "living habitat" that serves as a nutrient filter, provides shelter for fish and a variety of invertebrates, and provides physical substrate for invertebrates and algae (Thayer and Phillips 1977, Jewett et al. 1999, Dean et al. 2000, Bostrom et al. 2006). Eelgrass is a major primary producer in the marine nearshore (McConnaughey and McRoy1979) and because it is located in shallow water, is susceptible to oil spills and other human disturbances (Short and Wiley-Eschevaria 1996, Dean et al. 1998, Duarte 2002, Larkum et al. 2006, Short et al. 2006). Eelgrass is especially susceptible to dredging, anchor scars, and events that reduce light penetration into the water column such as runoff (increased turbidity) or nutrient addition (Walker et al. 1989, Oleson 1996, Hauxwell et al. 2003, Neckles et al. 2005, Terrados et al. 2006).

The purpose of this sampling is to assess changes in the extent of eelgrass over time. In this report, we examine results from sampling eelgrass cover in KATM, KEFJ, and WPWS. The sampling is designed to examine a portion of a large eelgrass beds (within beds of approximately 1 km^2) over temporal scales of several years.

Methods

We sampled the percent cover of eelgrass at up to five designated sites in each area from 2010-2122. All sampling was conducted in early summer when eelgrass beds generally have reached their seasonal maximum in extent and density of plants. All beds sampled were in sheltered bays and were at beds in closest proximity to the randomly selected rocky intertidal sites (see intertidal invertebrates and algae section).

At each site, we sampled eelgrass within a prescribed area along a shoreline of approximately 200 m in length. The width of each bed examined depended on the depth contour at each site, but was generally on the order of 50 to 100 m. The areas sampled were bounded by an approximately 200 m segment of shoreline over which eelgrass was observed and extended offshore to a distance approximately 15 m beyond the last observed eelgrass. The percent cover of eelgrass within this area was estimated by determining the presence or absence of eelgrass at approximately evenly spaced intervals along a series of transects running perpendicular to shore that were spaced approximately 20 m apart. Presence or absence at each observation point was determined using an underwater video camera lowered from a small inflatable boat and/or a single-beam sonar.

These surveys will allow us to detect changes in average density of eelgrass over time. While we do not know the types of changes that might occur, these might include local reduction in cover due to increased boating activity and associated anchor scars, a lowering of the upper depth limitation due to a decline in water clarity, or larger scale die offs due to diseases or contaminants.

Results

The percent of observations with eelgrass present ranged from 25% to 82% in 2012 across all three regions (Tables 1, 2, 3). The highest percent covers observed in KATM were at Amalik Bay in 2010 and 2012. The highest percent covers observed in KEFJ were Harris Bay in 2010 and Nuka Pass in 2011 and 2012. The highest percent covers observed in WPWS were at Johnson Bay in 2010 and 2011 and in Iktua Bay in 2012.
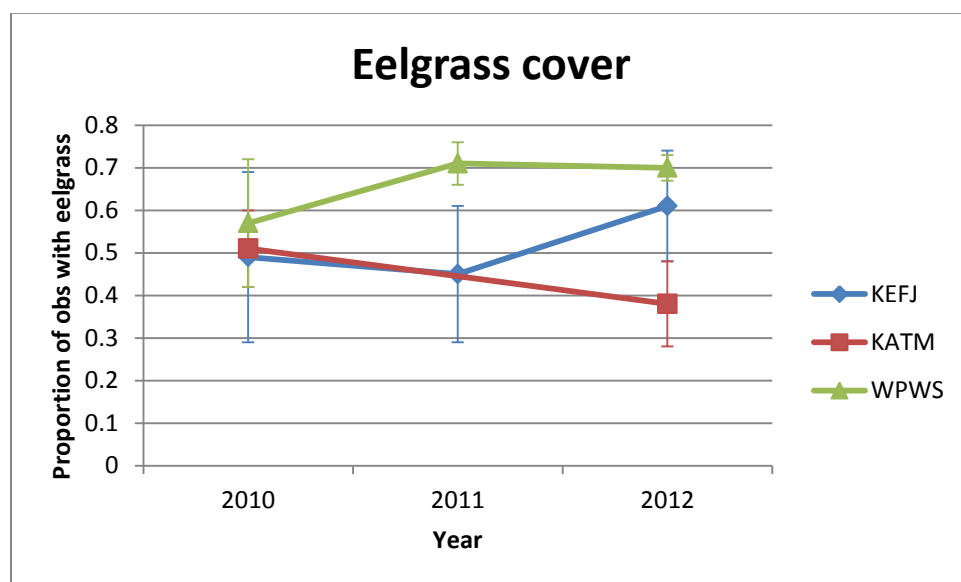
Figure 15. Proportion of observations with eelgrass present in KATM, KEFJ, and WPWS, 2008-2012. Error bars indicate 95% CI. KATM was not sampled in 2011.

Discussion

Using the methods briefly described above, we were able to estimate percent cover by eelgrass in designated sites. Data collected through 2012 should be sufficient to allow us to conduct power analyses to determine our ability to detect change in eelgrass cover over time; this will be considered as part of ongoing efforts.

Recommendations

Based on replicate sampling completed in 2008 (Coletti et al. 2009), our analysis indicated that the method produces relatively precise estimates of the relative density of eelgrass. We recommend the continuation of annual eelgrass bed sampling. Refinement of the SOP is on-going but we expect to finalize it in 2014.

Marine Bird Surveys

Introduction

Marine birds and mammals are important constituents of marine ecosystems and are sensitive to variation in marine conditions. Our focus on nearshore marine bird monitoring will be on species that are relatively abundant and trophically linked to the nearshore food web where the kelps and seagrasses contribute substantially to primary productivity and benthic invertebrates, such as clams, mussels and snails, transmit that energy to higher level trophic level fishes, birds and mammals. Species of focus in the nearshore food web include black oystercatchers (*Haematopus bachmani*), cormorants (*Phalacrocorax* spp.), glaucous-winged gulls (*Larus glaucescens*), black-legged kittiwakes (*Rissa tridactyla*), goldeneye ducks (*Bucephala* spp.) (winter density and distribution), harlequin ducks (*Histrionicus histrionicus*), pigeon guillemots (*Cepphus columba*), mergansers (*Mergus* spp.) and scoters (*Melanitta* spp.). Because other birds and mammals will be encountered in the course of monitoring nearshore species, observations of all marine birds and mammals are recorded.

The sea ducks and black oystercatcher were selected for focus because of their reliance on habitats and prey associated with nearshore marine communities. These species play an important role as top level consumers of nearshore invertebrates, including mussels, clams, snails, and limpets, that are being monitored under the intertidal invertebrates and algae component (Draulans 1982, Marsh 1986a and b, Meire 1993, Lindberg et al. 1998, Hamilton and Nudds 2003, Lewis et al. 2007). Therefore, understanding changes in the abundance of these bird species over time is an important metric for nearshore monitoring. Abundance estimates will be enhanced by the monitoring of nearshore invertebrates, which focuses on their prey populations. Moreover, monitoring trends in abundance of the various guilds of other marine birds (e.g. pigeon guillemots, black-legged kittiwakes, and cormorants) that utilize other food sources may improve the ability to discriminate among potential causes of change in seabird populations and the nearshore ecosystem. For example, concurrent changes in sea ducks, which forage on nearshore invertebrates, and the pigeon guillemots that forage on small fish, may suggest a common cause of change, one that may be independent of food. Such an approach may provide insights related to competing hypotheses relative to cause of change within or among populations (Petersen et al. 2003). In addition many of these species, including the harlequin duck, Barrow's goldeneye, and black oystercatcher, were impacted by the *Exxon Valdez* oil spill, and exhibited protracted recovery periods as a consequence of lingering oil in nearshore habitats in western Prince William Sound (Andres 1999, Trust et al. 2000, Esler et al. 2000a and b, Esler et al. 2002). Long-term monitoring of these species at different locations will likely provide increased confidence in assessment of the status of these populations relative to restoration and recovery from the 1989 spill. Additionally, existing data collected using comparable methods are available from other nearshore habitats in the Gulf of Alaska for periods up to 20 years (Irons et al.1988, Irons et al. 2000).

Methods

Standardized surveys of marine birds were conducted in KATM (2006-2010 and 2012) and KEFJ (2007-2012) between late June and early July. Similar surveys are conducted by the USFWS in WPWS, but results for that area are not reported here. Surveys are conducted from small vessels (5-8 m length) traveling at speeds of 8-12 knots along selected sections of coastline that represent independent transects. The transect width is 200 m and the boat represents the midpoint. Transects are surveyed by a team of three. The boat operator generally surveys the 100 m offshore area of the transect, while a second observer surveys the 100 m nearshore area. The third team member enters the observations into a laptop running program dLOG, specifically designed for this type of surveying, and assists with observations. All marine birds and mammals within the 200 m transect width are identified and counted. All transects considered in this analysis are run 100 m offshore and parallel to the shoreline. Detailed descriptions of methods and procedures can be found in the Marine Bird and Mammal Survey SOP (Bodkin 2011a).

The survey design consists of a series of transects along shorelines such that a minimum of 20% of the shoreline is surveyed. Transects are systematically selected beginning at a random starting point from the pool of contiguous 2.5-5 km transects that are adjacent to the mainland or islands, plus the lengths of transects that were associated with islands or groups of islands with less than 5 km of shoreline.

Each species is identified as important to nearshore food webs and as an important indicator of change (Dean and Bodkin 2011a, Dean et al. 2014). Several species were grouped into higher order taxa (e.g., cormorants, mergansers and scoters) because identification to species within these groups was not always possible. Cormorant species included pelagic, red-faced, and double crested cormorants. Merganser species include common merganser and red-breasted mergansers. Scoters included surf, black, and white-winged scoters.

Results

Only focal species densities and standard errors observed on nearshore transects are reported here (Figures 15-22). In general, there have been no notable shifts in densities over time. However, a possible decline in black oystercatcher densities has occurred in KATM, but more rigorous analysis is needed.
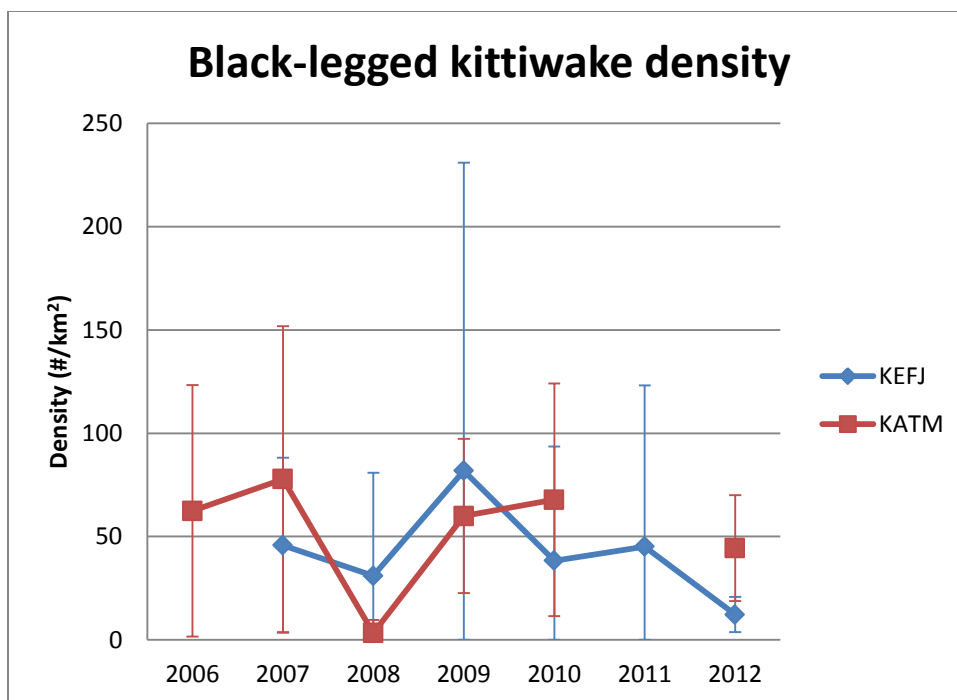
Figure 16. Density of black-legged kittiwake in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.
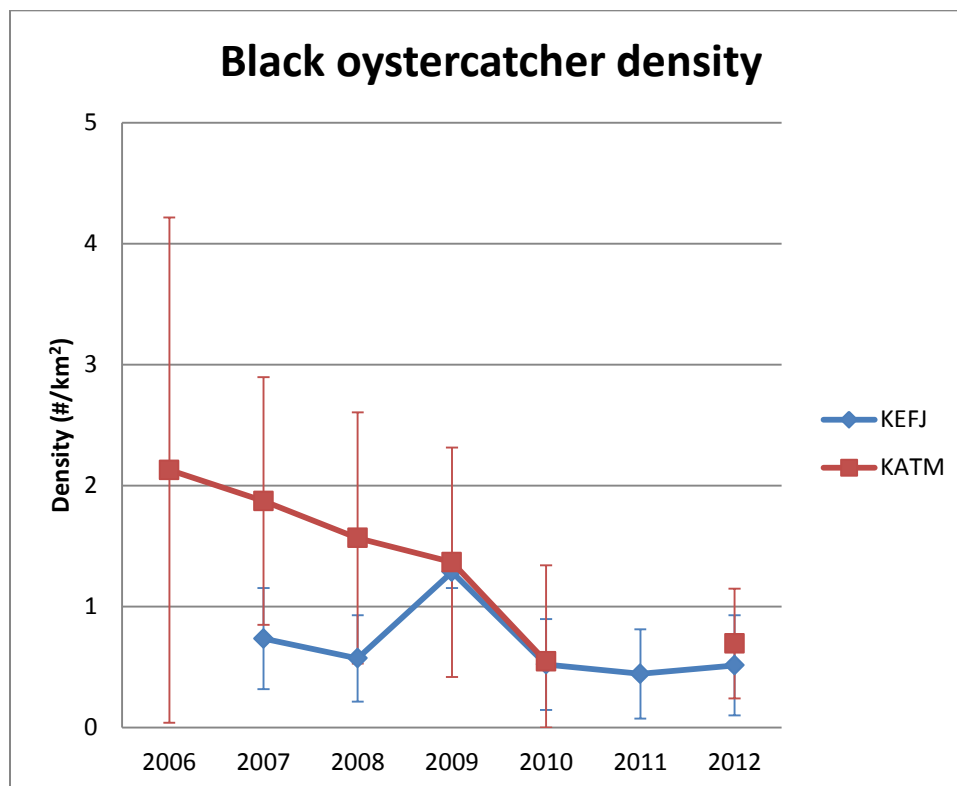
Figure 17. Density of black oystercatcher in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.

17
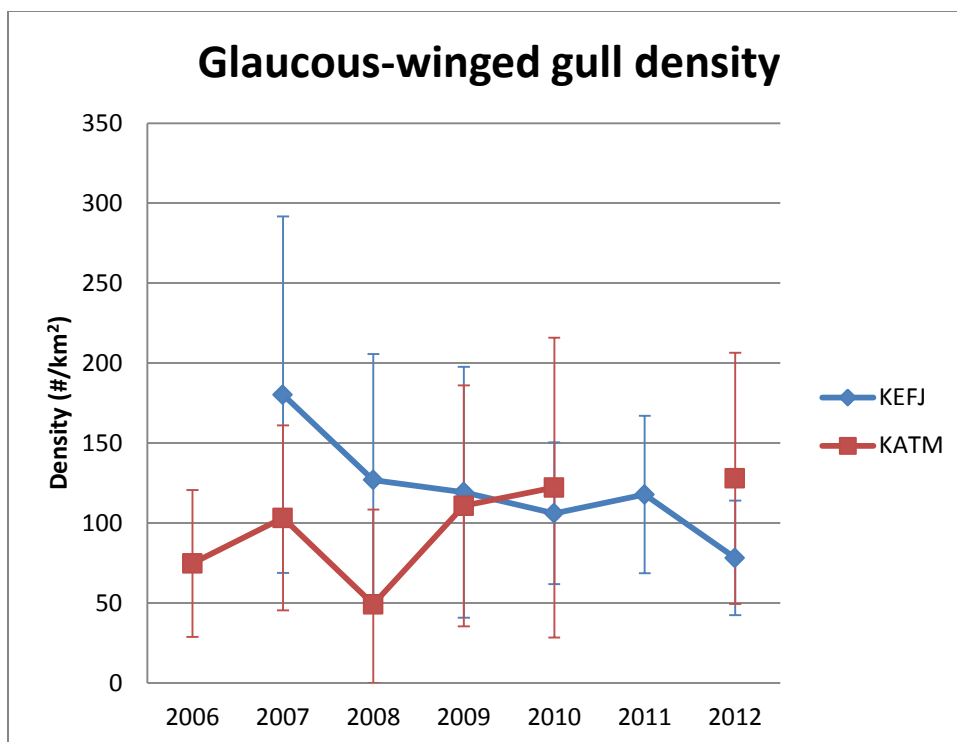
Figure 18. Density of glaucous-winged gull in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.
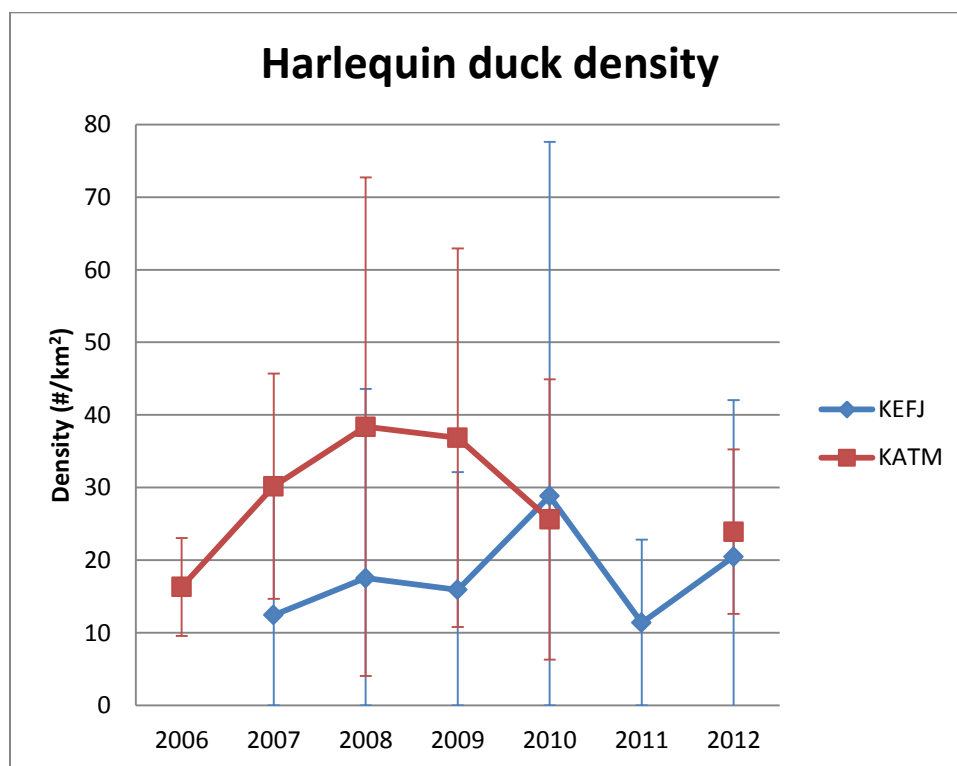
Figure 19. Density of Harlequin duck in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.
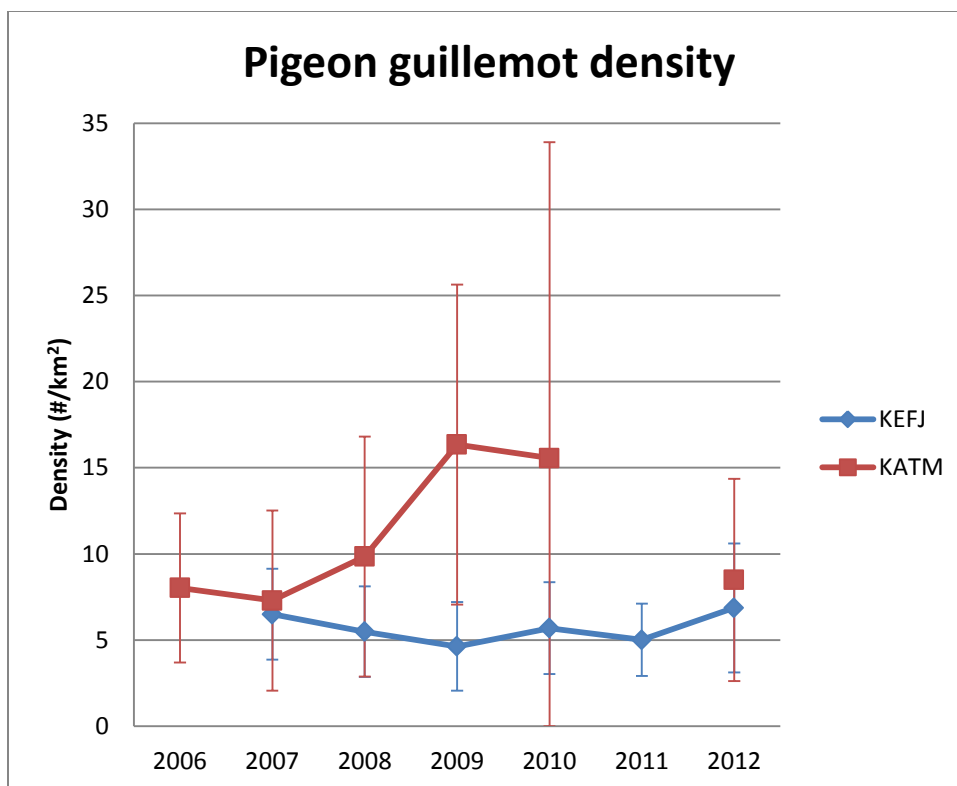
Figure 20. Density of pigeon guillemot in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.
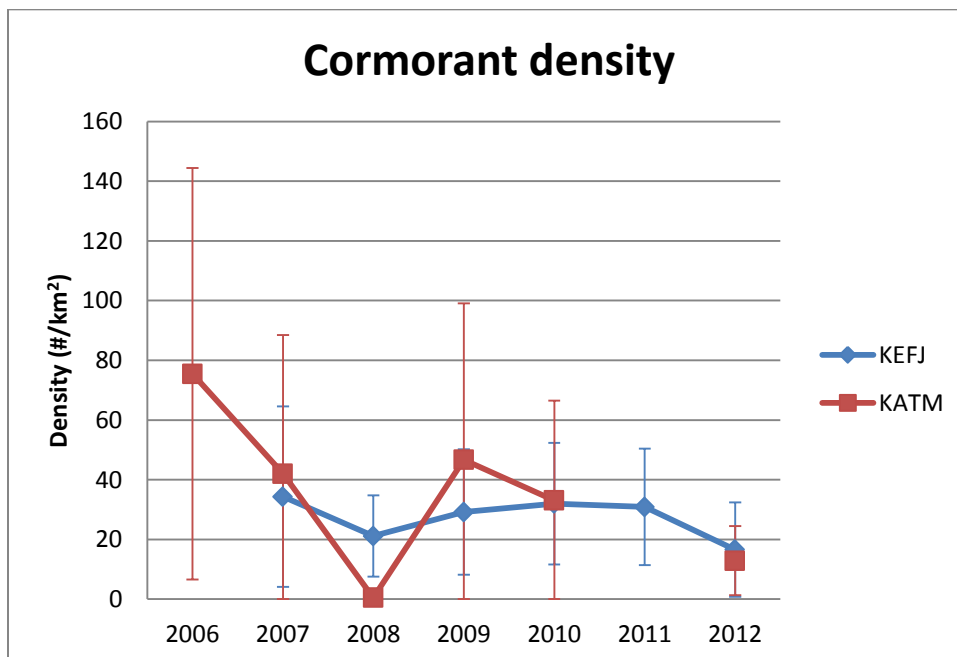
Figure 21. Density of cormorants in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.
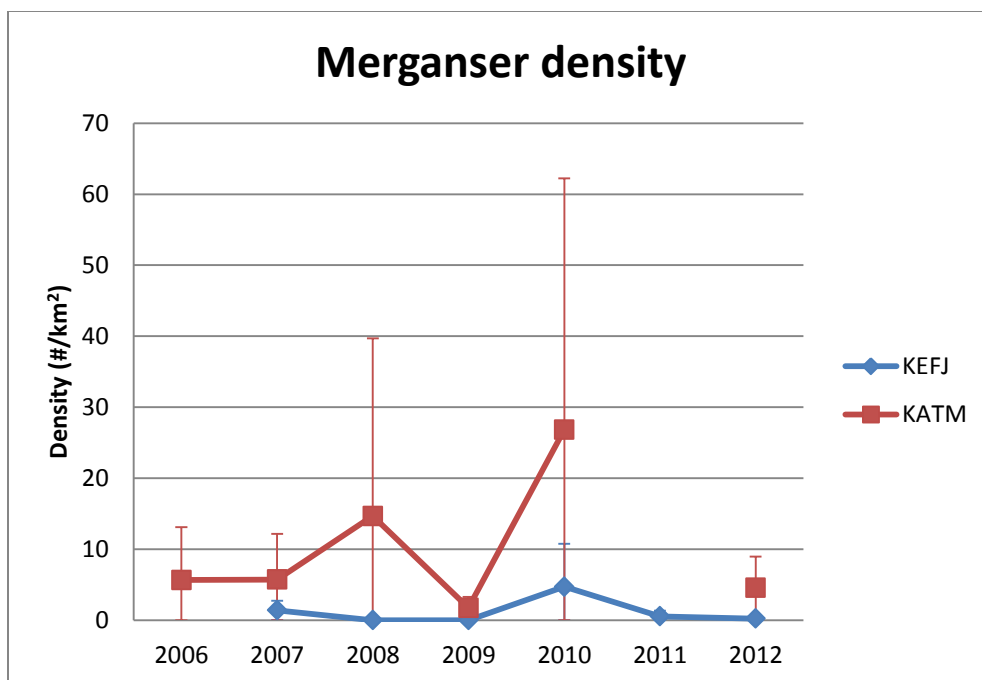
Figure 22. Density of mergansers in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.
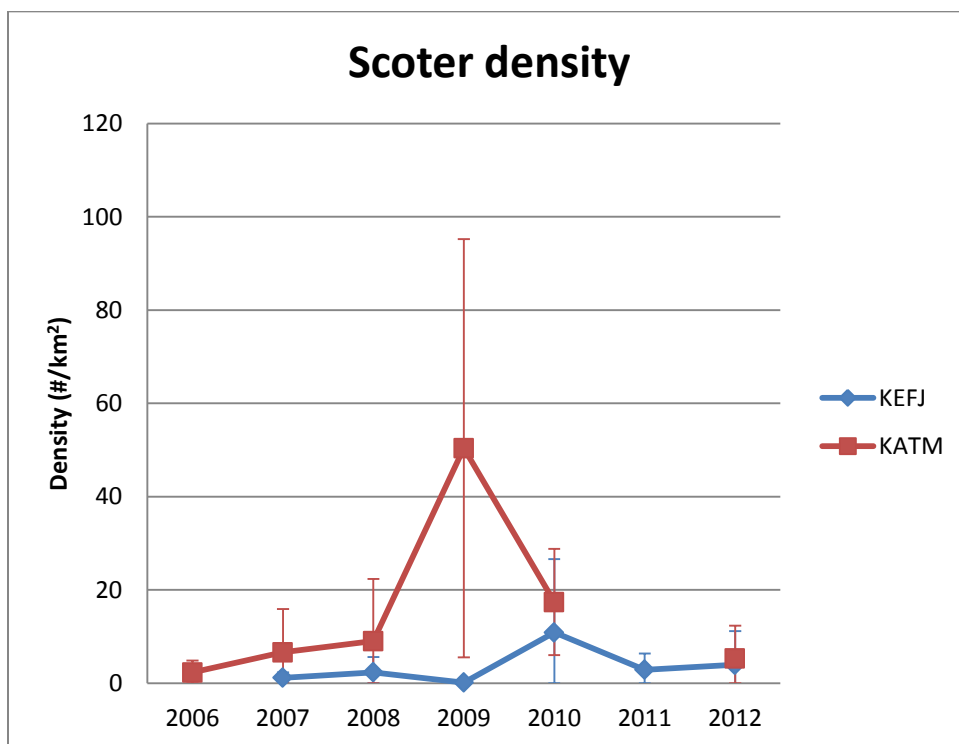
Figure 23. Density of scoters in KATM and KEFJ, 2006-2012. Error bars indicate 95% CI.

Discussion

KATM and KEFJ continue to be sampled annually during the summer. These shoreline skiff surveys provide baseline information on species composition, distribution and density for summer populations of marine bird and mammal fauna that occur in the nearshore waters of KATM and KEFJ. Because components of the marine bird and mammal fauna may change seasonally, inference of species composition, distribution, and densities to other seasons cannot be made. In particular, it is likely that some sea duck species that were rare or absent in the summer may be more common as over wintering residents (e.g. goldeneye, scoters, and long-tailed ducks). Sustainability of long-term monitoring programs requires the optimization of sampling intensity and efforts to minimize costs while concurrently having sufficient power to detect a trend. While there has been critical thought in the past regarding these questions, current available analytical methods now allow for the use of existing data to estimate number of samples and sample frequency required to detect a specified trend as well as examine effects contributing to variation, such as imperfect detection. An optimization exercise using existing data will occur in 2014.

Recommendations

We recommend that survey effort continue until further analysis can be completed. These datasets will be examined to determine levels of change that we can reasonably expect to detect based on this sampling method. We will also explore the possibility of re-allocating sampling efforts to specific habitat types or incorporate replicate sampling to enhance our ability to detect trends for species of interest.

Black Oystercatcher Sampling

Introduction

The black oystercatcher is a common and conspicuous member of the rocky and gravel intertidal marine communities of eastern Pacific shorelines and is completely dependent on nearshore marine habitats for all critical life history components including foraging, breeding, chick-rearing, and resting (Andres and Falxa 1995). During the late spring and summer breeding season pairs establish and defend both nest and forage areas, and these territories and nest sites can persist over many years (Groves 1984, Hazlitt and Butler 2001) with individual life expectancy exceeding 15 years (Andres and Falxa 1995). The diet consists primarily of mussels and limpets, which are ecologically and culturally important constituents of the intertidal community. The species is considered a Management Indicator Species by the Chugach National Forest and a species of concern nationally (Brown et al. 2001) and regionally (Alaska Shorebird Working Group 2000), and is widely recognized as a species representative of nearshore habitats. Because of their complete reliance on intertidal habitats, their reproductive biology, and foraging ecology, black oystercatchers are particularly amenable to long-term monitoring (Lentfer and Maier 1995, Andres 1998).

As a "keystone" species (Power et al. 1996), the black oystercatcher has a large influence on the structure of intertidal communities that is disproportionate to its abundance. The black oystercatcher receives its recognition as a keystone species through a three-trophic-level cascade initiated by the oystercatcher as a top level consumer in the nearshore (Marsh 1986a and b, Hahn and Denny 1989, Falxa 1992, Andres and Falxa 1995) whose diet consists largely of gastropod (limpets; *Lottia, Acmea*, and *Colisella* spp.) and bivalve mollusks (mussels; *Mytilus* spp.) that are ecologically important in the intertidal community. As a consequence of oystercatcher foraging, large numbers of herbivorous limpets can be removed (Frank 1982, Lindberg et al. 1987), resulting in shifts in limpet species composition and reduced size distribution (Marsh 1986a, Lindberg et al. 1987). As a consequence of reduced limpet densities and the diminished grazing intensity that results, algal populations respond through increased production and survival, resulting in enhanced algal populations (Marsh 1986a, Meese 1990, Wootton 1992, Lindberg et al. 1998). Additionally, like many other invertebrate, avian and mammalian predators in the nearshore, a large fraction of the oystercatcher's diet consists of mussels, an important filter feeding bivalve (Knox 2000, Menge and Branch 2001). Because the oystercatcher brings limpets, mussels and other prey back to its nest to provision chicks (Webster 1941, Frank 1982, Hartwick 1976, Lindberg et al. 1987), collections of those shell remains at nests provides an opportunity to obtain an independent sample of the species composition and size distribution of common and important nearshore invertebrate prey species that are directly estimated under intertidal algal and invertebrate vital signs (Intertidal Invertebrates and Algae section of this report). The collection of black oystercatcher diet and prey data offers a unique perspective into processes structuring nearshore communities (Marsh 1986a and b, Lindberg et al. 1987), including the potential consequences of anticipated increases in human presence and disturbance (Lindberg et al. 1998). Further, contrasting relative abundances and size-class composition of invertebrates collected under two independent protocols should increase our understanding of the processes responsible for change in nearshore ecosystems.

At a global scale, intertidal communities have been impacted by human activities (Liddle 1975, Kingsford et al. 1991, Povery and Keough 1991, Keough et al. 1993, Menge and Branch 2001) and one of the primary capabilities and intents of the nearshore monitoring program is to provide early detection of change in nearshore communities and to separate human from natural causes of change. Because of the critical nature of intertidal habitats for both breeding and foraging, black oystercatchers are particularly sensitive indicators to disturbances in the nearshore (Lindberg et al. 1998). Specifically, black oystercatchers nest exclusively in a narrow band just above the intertidal but below terrestrial vegetation, where eggs are laid in exposed nests consisting of depressions in pebbles, sand, gravel, and shell materials. During the 26-32 d incubation phase of reproduction, eggs are susceptible to predation by other birds (primarily C*orvids*; Lentfer and Meier 1995) and mammals (Vermeer et al. 1992), as well as human disturbance and trampling. Similar disturbance effects occur during the chick rearing stage, which lasts approximately 38 d (Andres and Falxa 1995). Thus, for several months during May-August, typically when humans are most present in nearshore habitats in Alaska, black oystercatchers are actively incubating or caring for young in a habitat with little protection from human induced disturbances. Chronic disturbance from human activities poses a significant threat to breeding black oystercatchers, either preventing nesting altogether, causing nest abandonment after eggs have been laid (Andres 1998), or through direct mortality of eggs or chicks. Monitoring of black oystercatcher density, breeding territory density and occupancy, and prey will provide a potentially powerful tool in identifying the magnitude and causes of inevitable change in Gulf of Alaska nearshore habitats and communities, particularly in response to the anticipated increased use and influence of those habitats by humans.

Methods
There are three components to the sampling related to black oystercatchers: estimation of breeding pair density and nest occupancy through oystercatcher-specific surveys; estimation of species composition and size distributions of prey returned to provision chicks; and density estimation of breeding and non-breeding black oystercatchers observed during the marine bird and mammal surveys. Results regarding the black oystercatcher density estimates are given in the marine bird survey section of this report. Detailed survey methods for estimation of nest occupancy and diet can be found in the black oystercatcher breeding territory occupancy and chick diet SOP (Bodkin 20011b). The detailed methods used to obtain marine bird densities can be found in the marine bird SOP (Bodkin 2011a) and in Bodkin et al. (2007b and 2008).

Black oystercatcher breeding territory density, nest occupancy, and prey data were collected along five 20 km transects, with each centered on the randomly (GRTS) selected rocky intertidal algal and invertebrate sites at KATM since 2006 (no sampling in 2011), KEFJ since 2007, and WPWS in 2007 and since 2010. Nest sites were located by surveying the shoreline in a small boat. All accessible nest sites were visited to determine the number of chicks and/or eggs present and all prey items (e.g. mussel or limpet shells) present at a nest site were collected. All prey were measured. Here, we present size data for most abundant prey species, Pacific blue mussels (*Mytilus trossulus*) and the limpets (*Lottia pelta*, *Lottia persona* and *Lottia scutum*).

Results

Density and Productivity

All five black oystercatcher GRTS transects were analyzed at the park / region level for nest density (nest/km) and productivity (chicks + eggs/nest) by year in KATM, KEFJ and WPWS. The mean density of active black oystercatcher nest sites at KATM ranged from 0.05 to 0.11 per km of shoreline from 2006-2012 (Figure 23). The mean density of active black oystercatcher nest sites at KEFJ ranged from 0.05 to 0.10 per km of shoreline from 2007-2012 (Figure 23) and from 0.06 to 0.14 per km of shoreline in WPWS between 2007 and 2012. The mean productivity (eggs + chicks / nest) ranged from 1.42 to 2.3 eggs + chicks / nest for KATM from 2006-2012 (Figure 24). The mean productivity (eggs + chicks / nest) ranged from 0.12 to 1.92 eggs + chicks / nest for KEFJ from 2007-2012 (Figure 24) and from 0.5 to 1.71 eggs + chicks / nest in PWS from 2007-2012. KEFJ and KATM showed a slight increase in the number of active nests in 2012. However, WPWS has showed a decline in the number of active nests since 2010 through 2012. KEFJ and WPWS have both shown a decline in the productivity of active nests, however, it appears that KATM has been stable from 2010-2012.
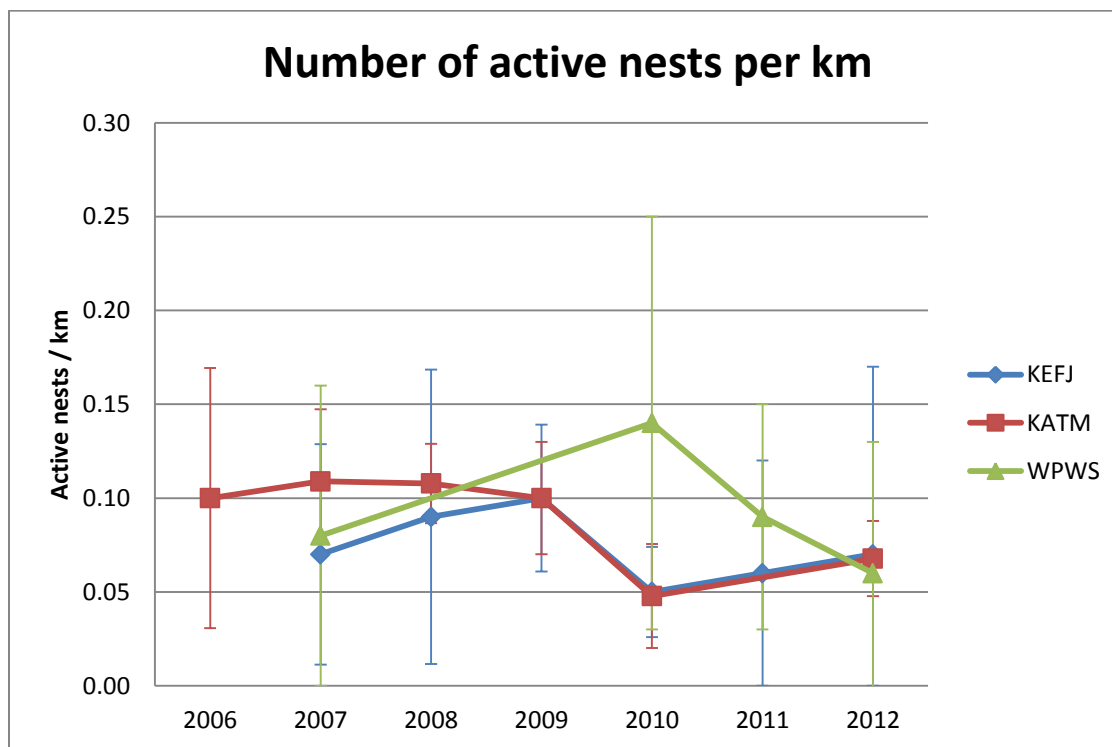
Figure 24. Number of active black oystercatcher nests / km in KATM, KEFJ, and WPWS, 2006-2012. Error bars indicate 95% CI.
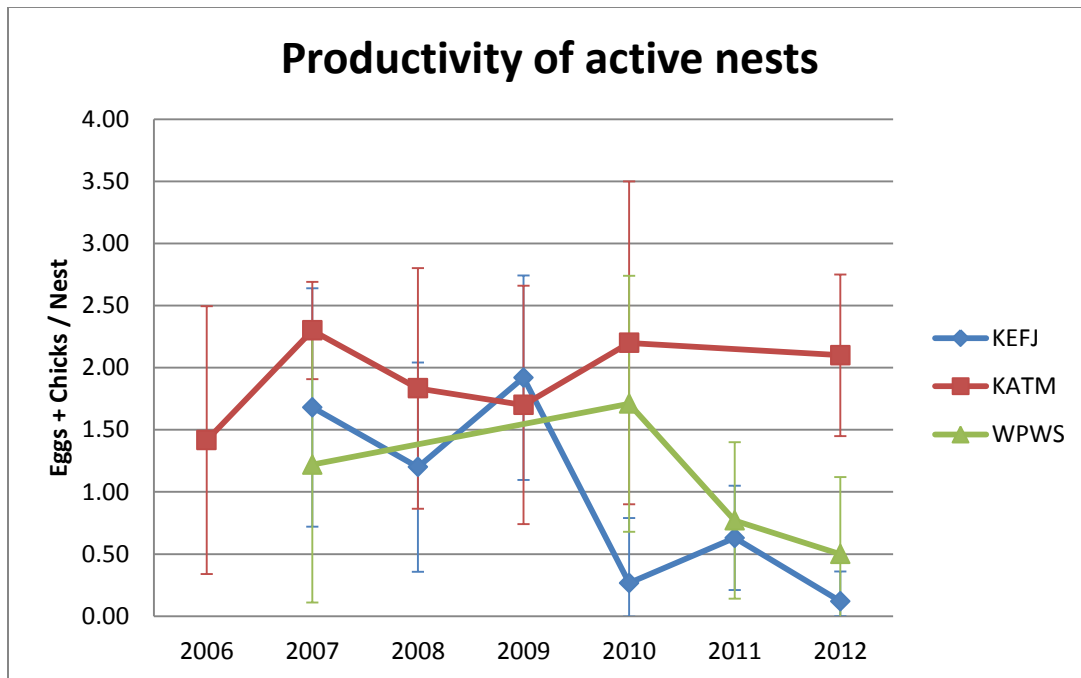
Figure 25. Productivity (eggs + chicks / nest) of active black oystercatcher nests / km in KATM, KEFJ, and WPWS, 2006-2011. Error bars indicate 95% CI.

Diet

Three species of limpets *(Lottia pelta, Lottia persona,* and to a lesser extent *Lottia scutum*) and the Pacific blue mussel *(Mytilus trossulus)* were the predominant prey items found at black oystercatcher nest sites in KATM, KEFJ and WPWS (Figures 25, 26 and 27). Together these species represented 94% of prey items found at KATM (2006-2012) nest sites and 96% in KEFJ and PWS (2007-2012) for all sampling years. No prey items were observed or collected in KEFJ in 2010 or 2012. Prey items were only available to be collected at two nests in KEFJ in 2011.
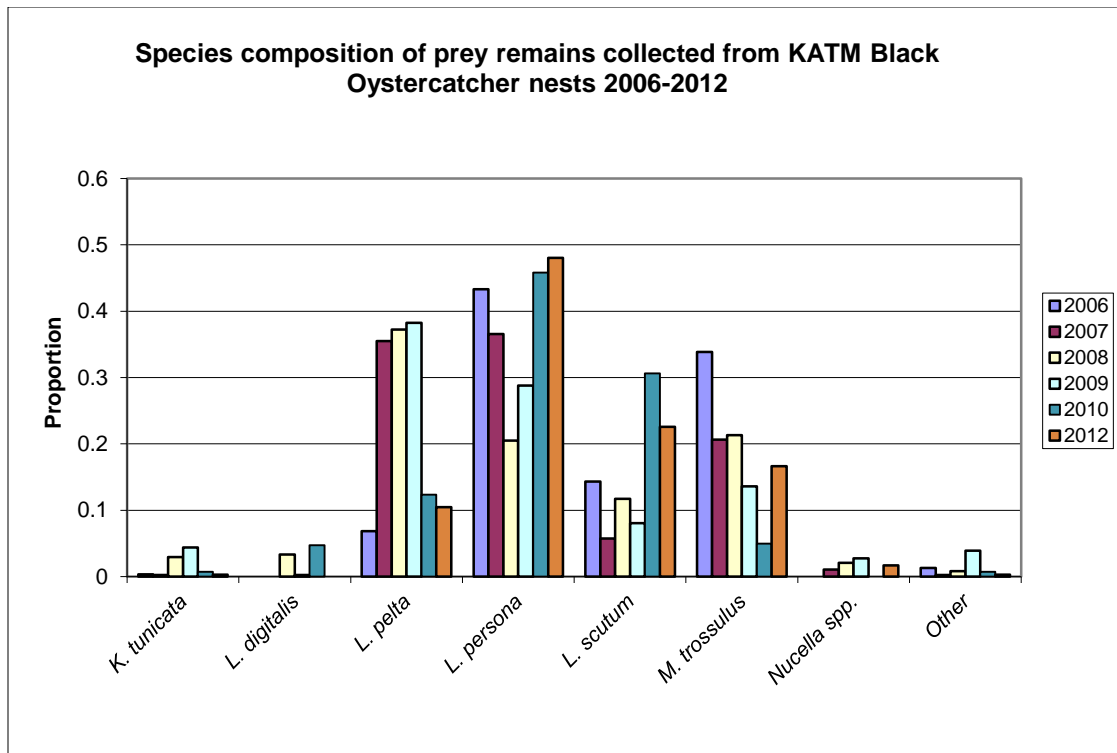
Figure 26. Species composition of prey items collected at active black oystercatcher in KATM, 2006-2012. KATM was not sampled in 2011.
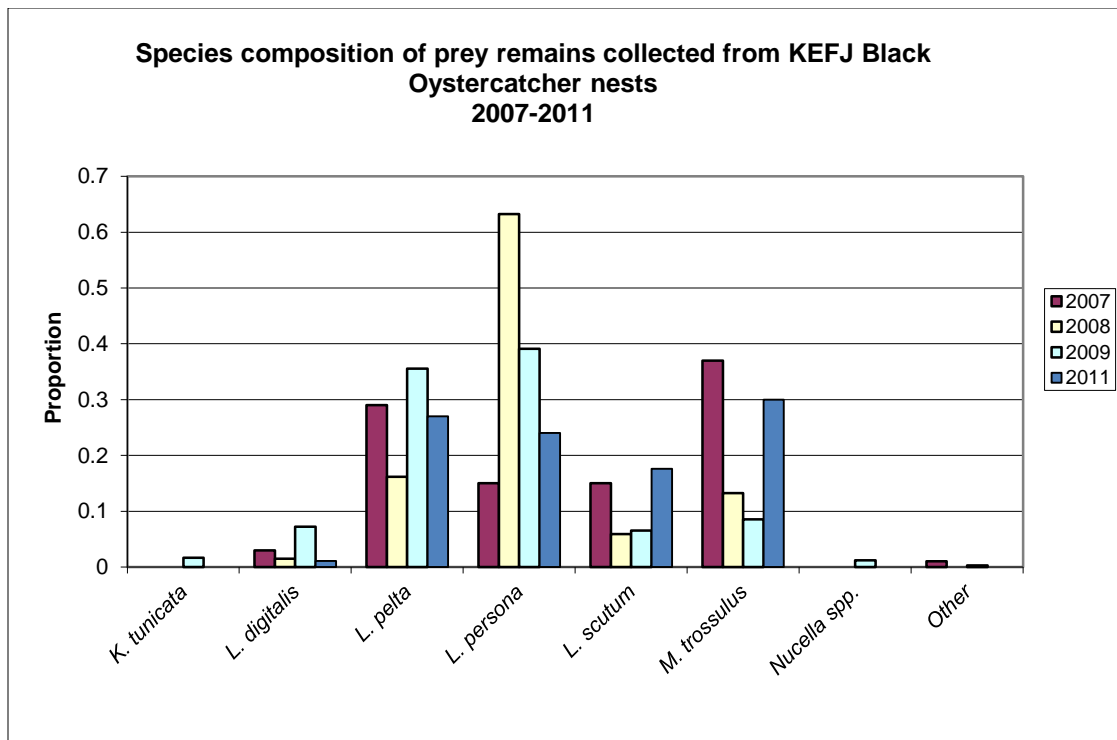
Figure 27. Species composition of prey items collected at active black oystercatcher in KEFJ, 2007-2012. No prey items were observed or collected in 2010 or in 2012.
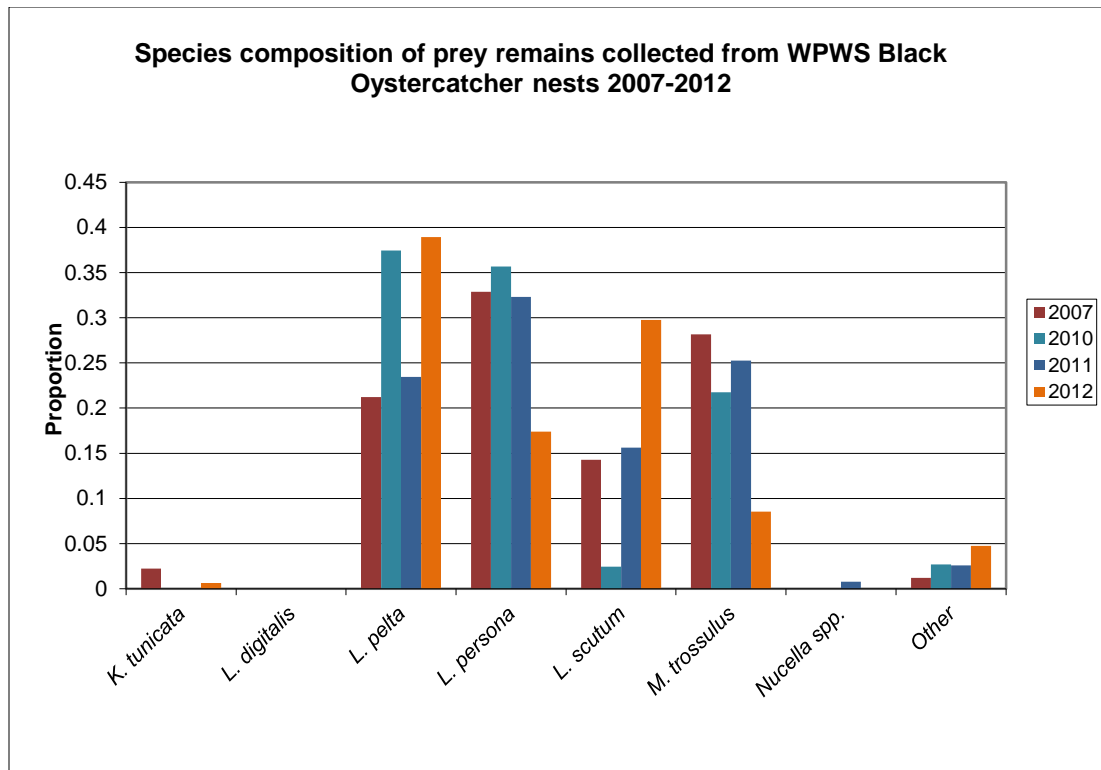
Figure 28. Species composition of prey items collected at active black oystercatcher in WPWS, 2007-2012. WPWS was not sampled in 2008 or 2009.

Prey size is measured for all species. However, we report only on the mean size of two of the most predominate species, the limpet *Lottia persona* and the mussel, *Mytilus trossolus*. Both of the species are also monitored for density and size within the sampling of Intertidal Invertebrates and Algae on Sheltered Rocky Shores SOP (Dean and Bodkin 2011b). Mean *L. persona* size ranged from 18.84 to 23.02 mm in KATM from 2006-2012, from 18.45 to 22.96 mm in KEFJ from 2007-2012 (no prey items observed in 2010 or 2012), and from 17.71 to 20.32 mm in WPWS from 2007-2012 (Figure 28). Mean *M. trossulus* size ranged from 27.44 to 45.05 mm in KATM from 2006-2012, from 20.07 to 29.92 mm in KEFJ from 2007-2012 (no prey items observed in 2010 or 2012), and from 30.57 to 35.27 mm in PWS from 2007-2012 (Figure 29).
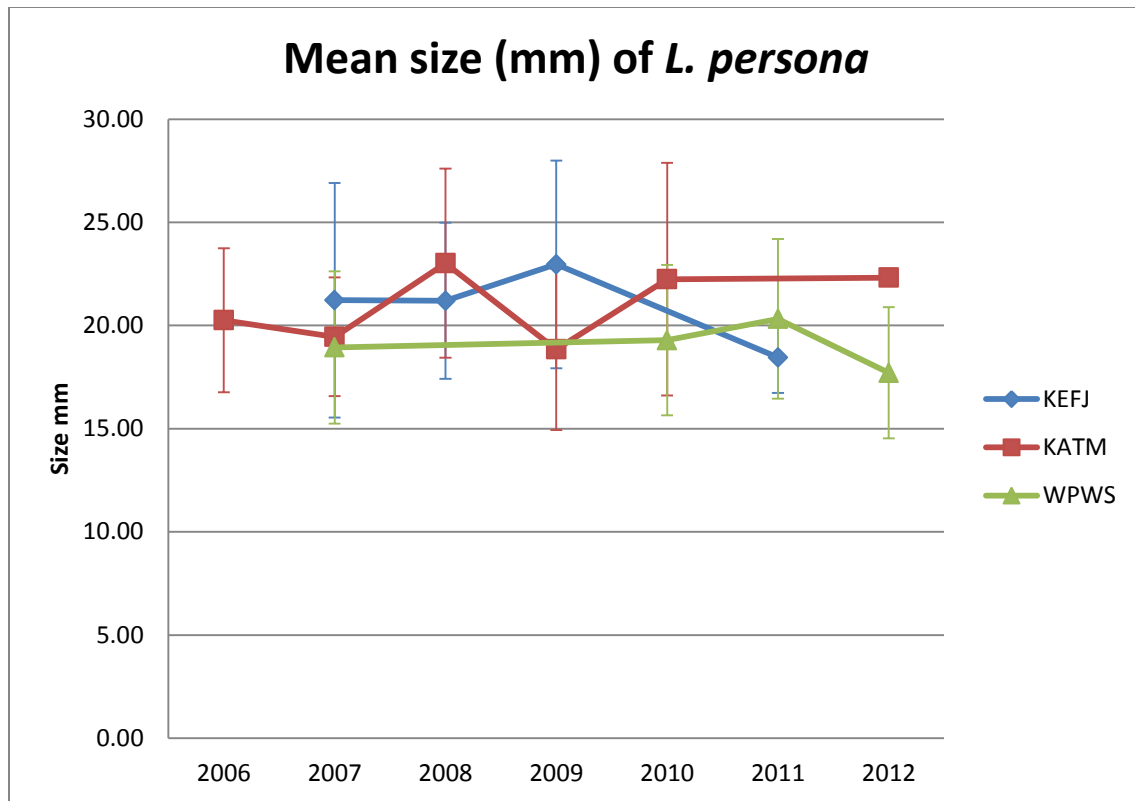
Figure 29. Mean size of *L. persona* from at active black oystercatcher nests in KATM (2006-2010, 2012), KEFJ (2007-2012) and WPWS (2007-2012). No prey items were observed in KEFJ in 2010 or 2012.
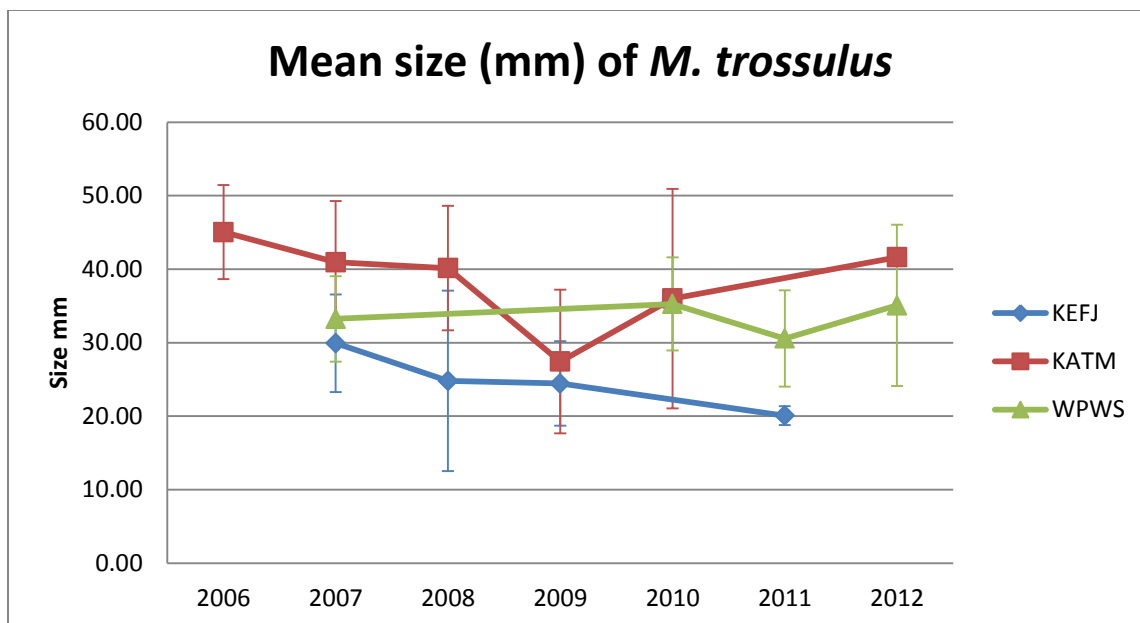
Figure 30. Mean size of *M. trossulus* from at active black oystercatcher nests in KATM (2006-2010, 2012), KEFJ (2007-2012) and WPWS (2007-2012). No prey items were observed in KEFJ in 2010 or 2012.

Discussion

Our data continues to show that black oystercatchers are targeting the larger size classes of mussels and limpets, based on our random sampling in the rocky intertidal and mussel bed sites. Variation in sizes of prey was generally relatively low. This is not surprising, but may be a key metric for monitoring purposes. Measurements of sea otter prey, pre- and post- arrival of sea otters in Glacier Bay, AK, have indicated a decline in prey sizes correlated with the increased occupation of Glacier Bay with sea otters (Bodkin et al. 2007a and c). A similar result possibly may occur as densities in nesting black oystercatchers changes. Lower densities of black oystercatchers may lead to increased densities of larger size classes of mussels and limpets sampled at the rocky intertidal sites and mussel beds or nest sites. The reverse may also be possible, with increased black oystercatcher densities associated with decreases in the densities of the larger size classes of prey.

Recommendations

Surveys of black oystercatcher abundance, nest density, and diet as reflected through prey remains brought to provision chicks have been successfully implemented in KATM, KEFJ and WPWS and have shown that at appropriate spatial scales of analysis, our data should continue to be collected with little revision. Sampling at the current intensity should allow us to detect trends in changes of nest density, productivity and diet (especially prey size) of the black oystercatcher. It appears as though breeding pairs may have multiple nests at a nest site and care should continue to be taken to recognize these as comprising the same nest site. It will be important to conduct future surveys as close as possible in time to these initial surveys and care must continue to be taken to minimize the disturbance to nests during sampling.

Sea Otter

Introduction

Sea otters (*Enhydra lutris*) are a common, conspicuous, and important component of the nearshore trophic food web throughout the North Pacific. They occupy all types of nearshore habitats from sheltered bays, estuaries, and fjords to exposed rocky coastlines (Kenyon 1969), but are constrained by their diving ability to habitats shallower than 100 m depth (Bodkin et al. 2004) and a near exclusive dietary reliance on benthic invertebrate prey (Riedman and Estes 1990). As a consequence of their nearshore distribution and relatively small home ranges, a rich literature exists on the biology, behavior, and ecology of the species. The sea otter provides one of the best documented examples of top-down forcing effects on the structure and function of nearshore marine ecosystems in the North Pacific Ocean (Kenyon 1969, VanBlaricom and Estes 1988, Riedman and Estes 1990, Estes and Duggins 1995) and are widely regarded as a "keystone" species in coastal marine ecosystems (Power et al. 1996). They cause well described top-down cascading effects on community structure by altering abundance of prey (e.g. sea urchins) which can in turn alter abundance of lower trophic levels (e.g. kelps). Sea otters generally have smaller home ranges than other marine mammals, eat large amounts of food, are susceptible to contaminants such as those related to oil spills, and have broad appeal to the public. From the mid-1980s through 2005 declines in sea otters have been observed in the Aleutian Islands (Doroff et al. 2003, Estes et al. 2005, Burn and Doroff 2005). As a result, the Western Alaska stock of sea otters, which occurs from Cook Inlet to the Western Aleutian Islands, which includes Katmai National Park and Preserve as well as Aniakchak National Monument and Preserve, was federally listed in September 2005 as threatened.

For the reasons outlined above, several metrics related to sea otters are incorporated under this vital sign. They include: observations of sea otter foraging, carcass collections to evaluate the age structure of the dying population, and aerial surveys to estimate population abundance. Because sea otters occur outside the boundaries of the skiff-based shoreline marine bird and mammal surveys, and because detection is not estimated during the skiff-based surveys, aerial surveys designed specifically to provide accurate and precise estimates of sea otter abundance (Bodkin and Udevitz 1999) are incorporated into the nearshore monitoring program.

Sea otter population abundance and trends are frequently influenced by the type and quantity of available prey (Kenyon 1969, Monson et al. 2000). Observations of foraging sea otters provide information on food habits, foraging success, (mean proportion of feeding dives that are successful) and efficiency (mean kcal/dive) based on prey numbers, types and sizes obtained by feeding animals. Because sea otter populations are often prey limited, data on foraging behavior will be useful in evaluating potential causes for differences in sea otter densities or trends among regions or years (Estes et al. 1982, 2003b, Gelatt et al. 2002, Dean et al. 2002, Bodkin et al. 2002, Tinker et al. 2008).

Due to high spatial variability in marine invertebrate populations (e.g. extreme patchiness) and difficulty in sampling underwater prey populations, foraging sea otters provide an alternative method to direct sampling of subtidal invertebrates. Following a successful foraging dive, sea otters return to the surface to consume their prey. This provides the opportunity to identify, enumerate, and estimate

the size of the benthic organisms they consume. Therefore sea otter foraging observations will provide data on species composition and sizes of subtidal invertebrate prey populations that are difficult to obtain directly. Observations collected over time may allow inference to changes in the species composition and sizes of the nearshore benthic invertebrate communities.

As a result of their nearshore distribution and relatively high density, moribund sea otters often haul out ashore, or their carcasses drift onto beaches. Annual collections of sea otter carcasses provide a record of the ages of dying individuals through analysis of cementum deposition in teeth (Bodkin et al. 1997). The age distributions of dying sea otters generated from annual carcass collections can provide a baseline against which future distributions can be compared and potentially provide inference regarding causes for change in population abundance, behavior, or diet (Monson et al. 2000, Estes et al. 2003a). Combined with data from a fresh carcass stranding program or annual population surveys, age-specific mortality data modeling can be used to inform managers regarding conservation decisions related to causes of mortality (Gerber et al. 2004, Tinker et al. 2006).

Brief summaries of the methods and 2012 results for sea otter foraging observations, carcass collections, and cementum tooth age analysis are presented in this report. The methods and results of the 2012 sea otter aerial survey in KATM are also reported here.

Methods
Aerial survey
The survey follows protocols described in detail in Bodkin and Udevitz (1999) which is summarized here. The survey is conducted from a small, single engine, float-equipped aircraft with both the pilot and observer able to observe out each side of the aircraft. The airplane is flown at a speed of 105 kph (65 mph) and at an elevation of 91 m (300 ft). The survey design consists of systematic sampling of 400 m wide transects spanning the survey area. Sampling intensity is proportional to expected sea otter abundance with most survey effort taking place where higher densities of sea otters are generally observed. The high density sea otter stratum extends from shore to 400 m seaward or to the 40 m depth contour, whichever is greater. Bays and inlets less than 6 km wide are also categorized as part of the high density stratum, regardless of depth. The remaining survey effort is over deeper, offshore waters where lower densities are generally observed (Figure 1). Specifically, the low density sea otter stratum extends from the high density stratum line to 2 km offshore or from the 40 m depth contour to the 100 m depth contour, whichever is greater. Intensive searches, initiated by the observer, are periodically conducted within the transect swaths to estimate the proportion of sea otters not initially detected during the strip counts. Strip counts are adjusted for the area not surveyed and by a detection correction factor to obtain an adjusted population size estimate. Additionally, groups larger than approximately 20 individuals are circled until a complete count is obtained and are treated as a separate stratum, uncorrected in the analysis. Surveys are conducted in alternate years in different areas; the 2012 survey was at KATM.

Survey transects used in KATM in 2012 were identical to those used in 2008 with spacing of transects in the shallow water strata or high density strata every 1.2 km and 2.4 km between transects in the deep water stratum or low density stratum. A total of 267 transects, representing approximately 975 linear km in the shallow water and deep water strata, were surveyed. The survey area ranged

from Cape Douglas to the southwest end of Cape Kubugakli at the park boundary. Transects located near river mouths were often dry at lower tides and thus not flown. In those circumstances, the transect was marked as such and removed from the area surveyed for analysis.

During the survey, we entered data using a custom survey application (Doug Burn, U.S. Fish and Wildlife Service, Anchorage, AK) in ArcPad (ESRI Inc., Redlands, CA). After the survey, data were post-processed in ArcMap (ESRI Inc.) and SAS 9.3 (SAS Institute, Cary, NC) to estimate population size.

Foraging

Prey composition, foraging success rate, and prey size were obtained from shore based observations of randomly selected foraging otters. Shore-based observations limited data collection to sea otters feeding within approximately 1 km of shore. High powered telescopes (Questar Corp., Hew Hope, PA) and 10X binoculars were used to record prey type, number, and size class during foraging bouts of focal animals. A bout consisted of observations of repeated dives for a focal animal while it remains in view and continues to forage (Calkins 1978). Assuming each foraging bout records the feeding activity of a unique individual, bouts were considered independent while dives within bouts were not. Thus the length of any one foraging bout was limited to 20 dives, or one hour, after which a new focal animal was chosen. Within each bout sampled the following data were recorded: date, start and end time, age class, sex, pup status and location coordinates. Foraging data collected include dive and surface interval times, success, prey species, number and size, and if prey were given or taken (typically given to a pup, or taken by a con-specific). The sampling design included the acquisition of foraging data within a 10 km radius of each of the five established rocky intertidal invertebrate and algal sites. The objective was to annually obtain data from 10 individuals within each of these 10 km buffers, a total of 50 bouts per year.

Sea otters in the study areas were generally not individually identifiable. In addition, some foraging areas may have been used more than others by individuals and by otters living in the area in general. Therefore individual sea otters may have been observed more than once leading to potential bias toward individuals sampled more than once. To minimize this potential, observers use characteristics such as sex, sizes, coloration, and pup presence to identify individuals. If more than one animal was observed foraging, selection was based on proximity, alternating between closest and furthest.

Carcass

Throughout the study areas in KATM, KEFJ, and WPWS we have identified segments of shoreline or offshore islands to search for sea otter carcasses. Annually, these areas have been consistently searched by two or more observers. Search patterns cover from the storm strand line to the water's edge and focus on areas where larger amounts of debris collect. When a carcass is found the skull and baculum, whiskers, and tissue if present, are collected. The following data are recorded: date, observers, condition of carcass, sex, parts collected, latitude/longitude, location on beach (e.g. strand line, above high tide, etc.), and cause of mortality (usually not known). A premolar tooth (or substitute if the premolar is not available) is sent to Matson's Laboratory in Montana for cementum layer age analysis.
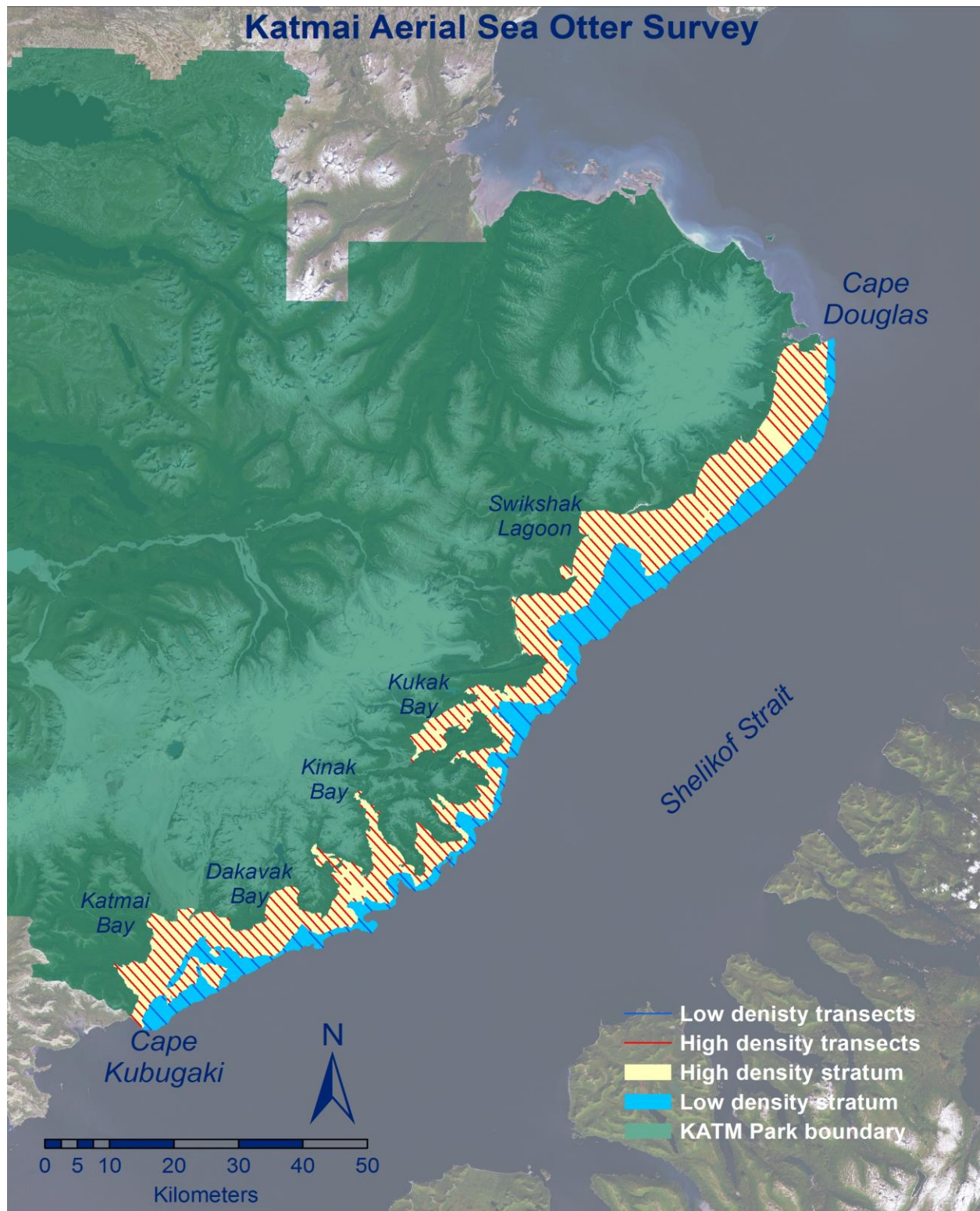
Figure 31. Sampling transect locations for the high density stratum (in yellow) and low density stratum (in blue) used in the aerial survey of sea otter abundance in KATM during August of 2012.

One of the objectives for this monitoring program is to detect levels of change deemed ecologically important (Dean and Bodkin 2011a, Dean et al. 2014). Ecologically relevant changes in sea otter population estimates have been set at 0.40 (40% increase or decrease). For the sea otter foraging data we have established a 0.35 change in the proportion of dominant prey categories, a 0.50 change in prey size and a 0.20 increase or 0.33 decrease in the number of hours needed to meet energetic requirements as ecologically relevant changes to detect. Programming capable of providing variance estimates of energy recovery rates is presently in revision, precluding power analysis for this metric. Power analysis for linear regression (Gerrodette 1993) was used to evaluate levels of change in focal species densities that could be detected over time. Forage data are analyzed at the spatial scale of a park. Future analyses may include finer spatial resolution analyses as sample sizes increase within each of the five buffers associated with the intertidal sites and should include caloric recovery rate power analyses. Grouping sea otter ages into juvenile, prime-age adult, and aged adult categories will be used to evaluate change over time. A 0.4 change in any age class has been determined to be ecologically significant.

Results

Aerial survey

Between 8 and 10 August, 2012, 267 transects in the high and low density strata were surveyed in KATM to estimate sea otter abundance (Table 1). The high density stratum consisted of approximately 985 km^2 and the low density of approximately 465 km^2, representing 975 linear km in the shallow water and deep water strata (Figure 31). One hundred and ninety seven transects comprising 793 km of high density transect length, and 70 transects comprising 182 km of low density transect were surveyed. Sea otters were observed on both high and low density transects (Table 1, Figure 32). Pups were primarily observed throughout the high density stratum and eleven large groups ($\geq$20) were also observed on transect, primarily in the high density stratum (Figure 32). The estimated detection probability was 0.53 resulting in a correction factor of 1.88 and a total estimated population size of approximately 8,644 (SE=1,243) sea otters residing within the surveyed area of KATM. The overall density is 5.96/km^2 (Table 1). Six large groups ($\geq$20) and one small group (n=1) of sea otters were observed off transect. Three of the large groups were in the vicinity of Dakavak Bay, and one each in Katmai Bay, Swikshak and off Cape Douglas. The one small group off transect was observed in Hallo Bay (Figure 33).

Table 1. Sea otter population abundance estimates for KATM from 2012. Uncorrected population size is the population size before the correction factor is applied to calculate adjusted population size.

Stratum	Uncorrected Population Size	Correction Factor	Adjusted Population Size	SE	Complete Counts	Corrected Density #/km2
High Density	3631	1.88	6808	1053	270	6.91
Low Density	256	1.88	479	197	81	1.03
Total			8644[1]	1243		5.96

[1] Total adjusted population size does not equal the sum of the high and low density strata because the adjusted population size also includes complete counts. Complete counts are treated as a separate stratum in the analysis.

Figure 32. Distribution and relative abundance of adult and pup sea otters in KATM, August 2012.

Figure 33. Groups of sea otters observed off transect, KATM, August 2012.

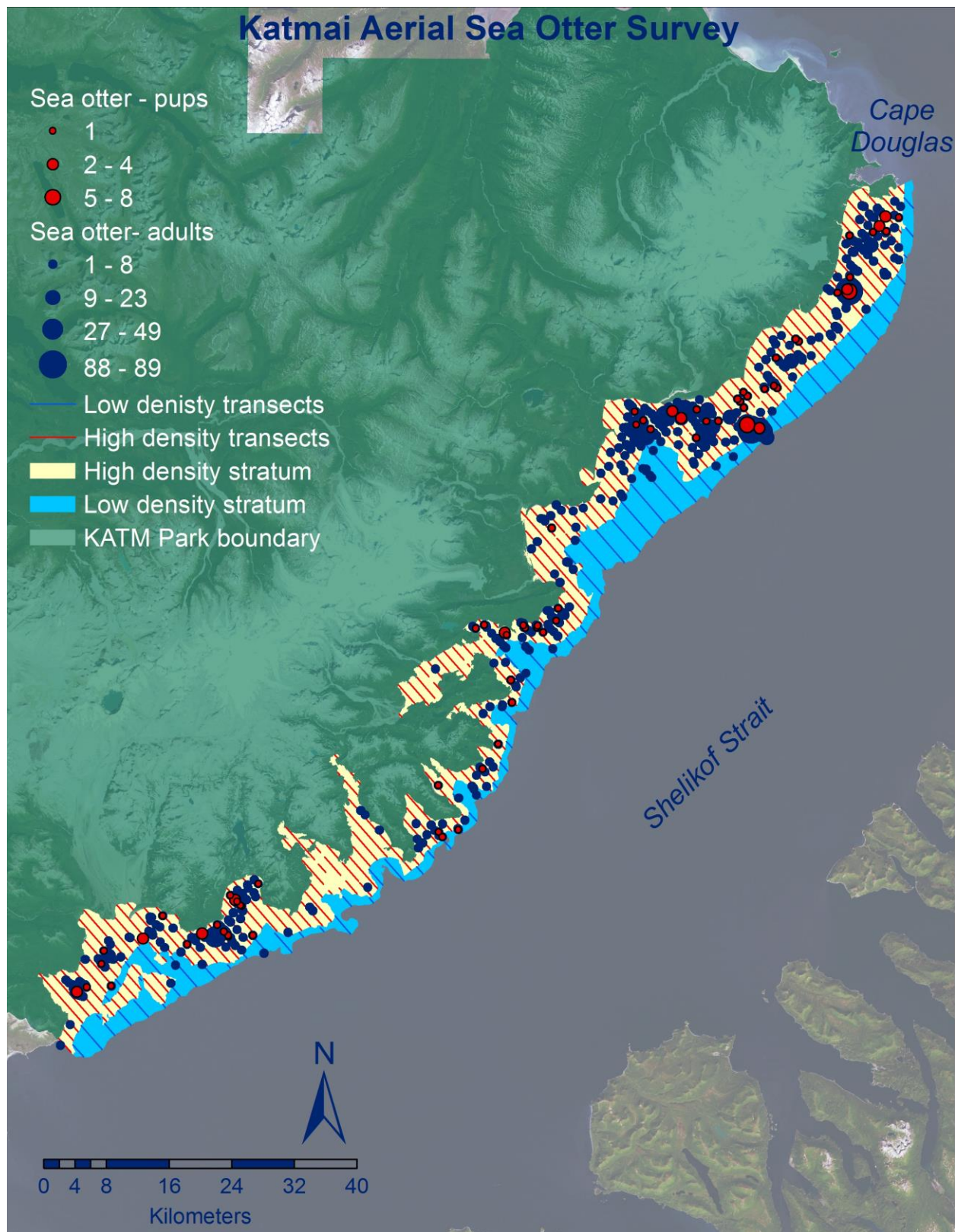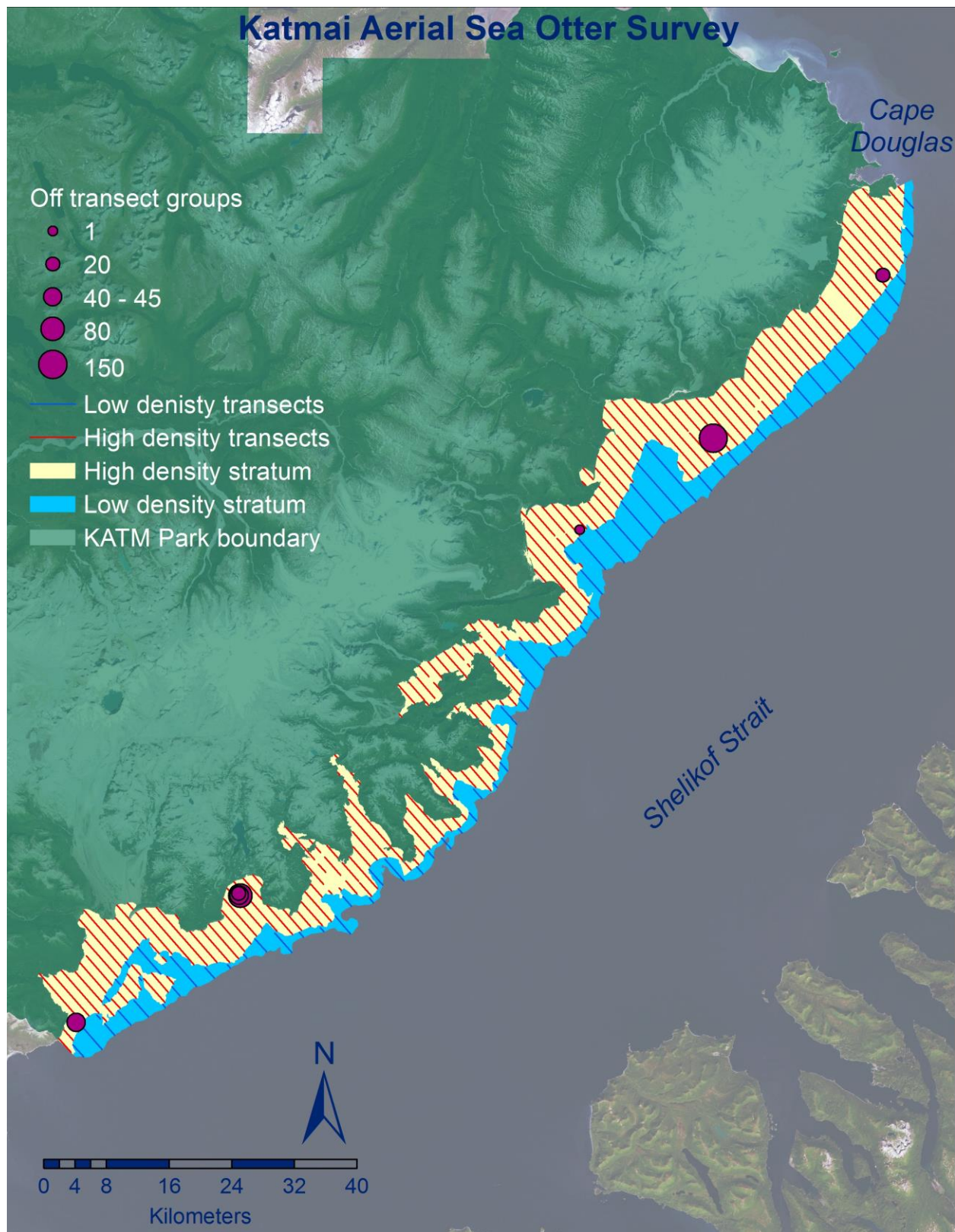**Foraging**

During six field seasons (2006-2010, 2012) at KATM we obtained data from 289 independent sea otter foraging bouts, consisting of 2,726 dives (Table 5). The prey recovery success rate was 88% for dives with known outcomes (range 82% - 92%) (Figure 33). During six field seasons (2007-2012) at KEFJ we obtained data from 283 independent sea otter foraging bouts, consisting of 2,509 dives (Table 5). The prey recovery success rate was 88% for dives with known outcomes (range 68% - 96%) (Figure 33). During five field seasons (2007, 2008, 2010-2012) at WPWS we obtained data from 427 independent sea otter foraging bouts, consisting of 2,372 dives (Table 5). The prey recovery success rate was 89% for dives with known outcomes (range 88% - 92%) (Figure 33).

Since 2006, we have observed sea otters feeding on at least 40 different prey items including bivalves, decapod crustaceans, gastropods, echinoderms, and fish. At KATM, clams dominated sea otter diets across all years of data collection, comprising approximately 60% numerically, (range 36% - 71%) of the diet (Figure 34). Chitons, crabs, mussels, octopus, snails, sea stars, sea urchins, and other prey each comprised less than 10% of the of prey recovered during most years, although exceptions do exist. In 2006 octopus accounted for 13% of identified prey, in 2008 chitons were 17%, in 2009 snails and urchins accounted for 30% and 14%, respectively, in 2010 snails accounted for 11%, and in 2012 snails and urchins accounted for 11% each of the prey retrieved.

At KEFJ, mussels (*Mytilus trossulus*) dominated sea otter diets across all but the most recent year of data collection, comprising 58% (range 27% - 79%) of the diet (Figure 34). In all years but 2012, clams were the second most prominent prey item comprising 28% (range 13% - 61%) of the diet. Otherwise, chitons, crabs, octopus, snails, sea stars, sea urchins, and other prey each comprised less than 10% of the of prey recovered.

At WPWS, clams dominated sea otter diets across all years of data collection, comprising 56% (range 47% - 96%) of the diet (Figure 34). In 2007 and 2010 mussels accounted for 30% and 23% of identified prey, in 2012 crab accounted for 16% of identified prey, while in 2011 and 2012 'other prey' accounted for 16% and 10%, respectively. The 'other prey' category for this location and this range of dates is comprised of non-clam/non-mussel bivalves, sea cucumbers, egg cases, and worms. This category should not be confused with 'unknown' prey which is not included in the prey composition calculations but will be shown in the mean size figures.

Sizes of prey captured by foraging sea otters vary by species (Figure 35). In KATM, the predominant prey, clams, averaged 54 mm over all sites and all years combined. Crabs (45mm), snails (32 mm), mussels (42 mm), urchins (39 mm), and unidentified (34 mm) prey items were smaller than the clams being retrieved while chitons (71 mm), and stars (99 mm) were larger than the clams.

In KEFJ, the predominant prey, mussels, averaged 24 mm over all sites and all years (Figure 35). Clams averaged 49 mm, crabs 60 mm, urchins 40 mm, and unidentified prey items were 27 mm. Sample sizes were low for the other prey categories.

In WPWS, the predominant prey, clams, averaged 48 mm over all sites and all years (Figure 35). Mussels averaged 23 mm, crabs 53 mm, urchins 34 mm, and unidentified prey items were 33 mm. Sample sizes were low for the other prey categories.

For clams and mussels, mean size per year has been reported in Figure 36. There is no observed difference in the size of clams between parks nor across years. Mussels, the primary prey item in KEFJ, have a similar mean size across years in KEFJ. Data are too scant to determine if the larger average size at KATM is meaningful. Unidentified prey size at both parks does not vary much across years. In KATM unidentified prey are consistently smaller than the mean size of the predominant prey while in KEFJ unidentified prey are similar in size to the predominant prey.

Table 2. Summary of sea otter foraging observations in KATM, KEFJ, and WPWS from nearshore monitoring data collection, 2006 - 2012. Foraging data were not collected in KEFJ and WPWS in 2006, WPWS in 2009, and KATM in 2011. A bout is the sampling unit for data analysis.

Year	Number of bouts observed			Number of dives observed			Mean number of dives per bout (SE)		
	KATM	KEFJ	WPWS	KATM	KEFJ	WPWS	KATM	KEFJ	WPWS
2006	65	.	.	451	.	.	6.7(0.2)	.	.
2007	54	45	81	498	471	365	7.7(0.2)	8.9(0.3)	7.6(0.3)
2008	38	57	5	427	392	28	8.6(0.3)	5.7(0.2)	6.3(1.0)
2009	36	37	.	392	269	.	8.4(0.3)	7.2(0.3)	.
2010	49	57	96	522	497	581	7.7(0.2)	7.0(0.3)	8.6(0.3)
2011	.	54	101	.	581	585	.	8.3(0.2)	7.9(0.3)
2012	47	33	144	436	299	813	7.5(0.3)	8.1(0.3)	8.1(0.2)
All Years	289	283	427	2,726	2,509	2,372	7.7(0.1)	7.6(0.1)	8.1(0.1)
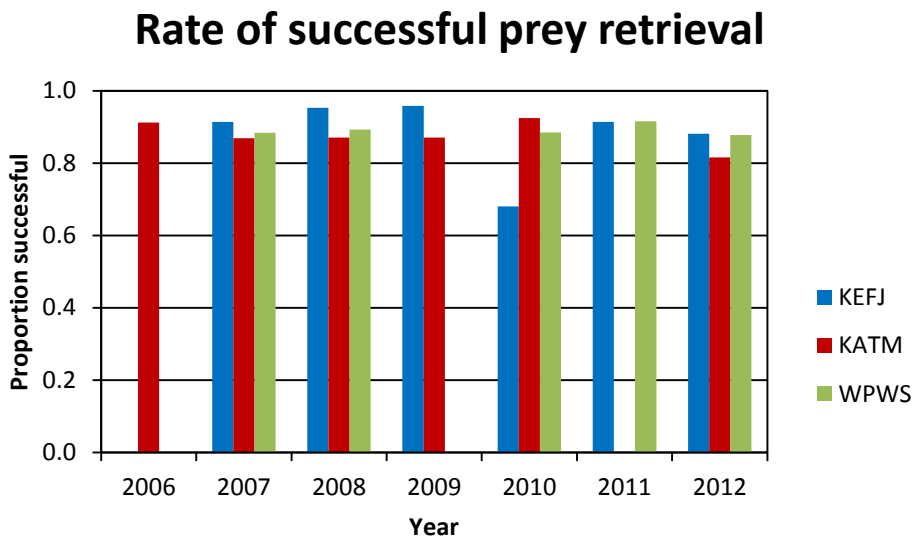
Figure 34. Success rate equals the proportion of known outcome dives where prey was successfully retrieved (Yes) by foraging sea otters in KATM 2006-2010, 2012, KEFJ, 2007-2012, and WPWS 2007-2008, 2010-2012. Dives in which otters were retrieving a previously collected prey item that had been dropped were not included. Additionally, a dive is only counted towards the success rate once, even if more than 1 item was retrieved.

Proportion clams in sea otter diets
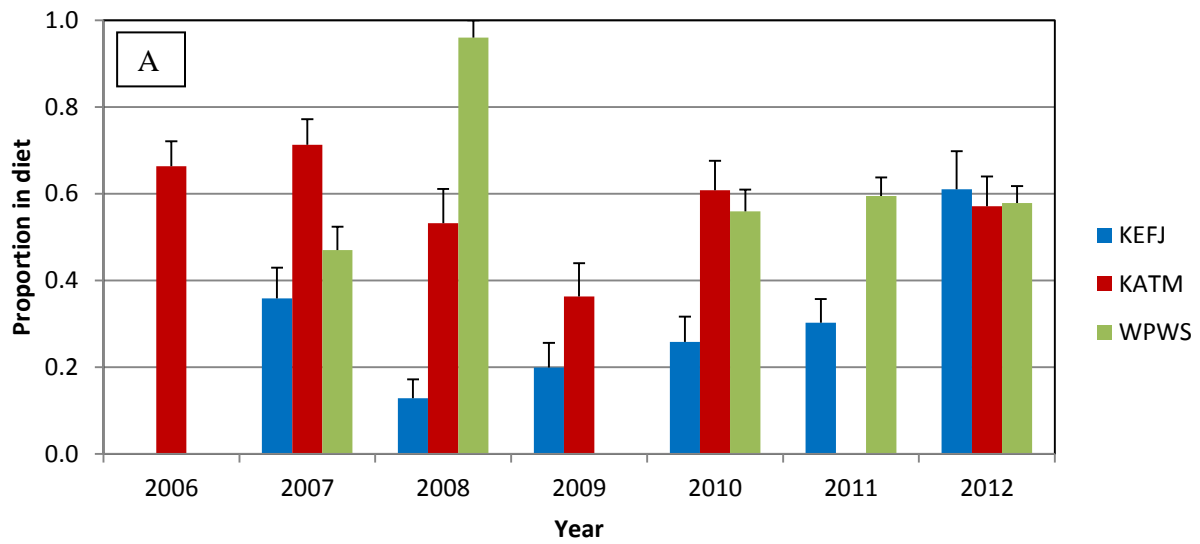
Proportion mussels in sea otter diets
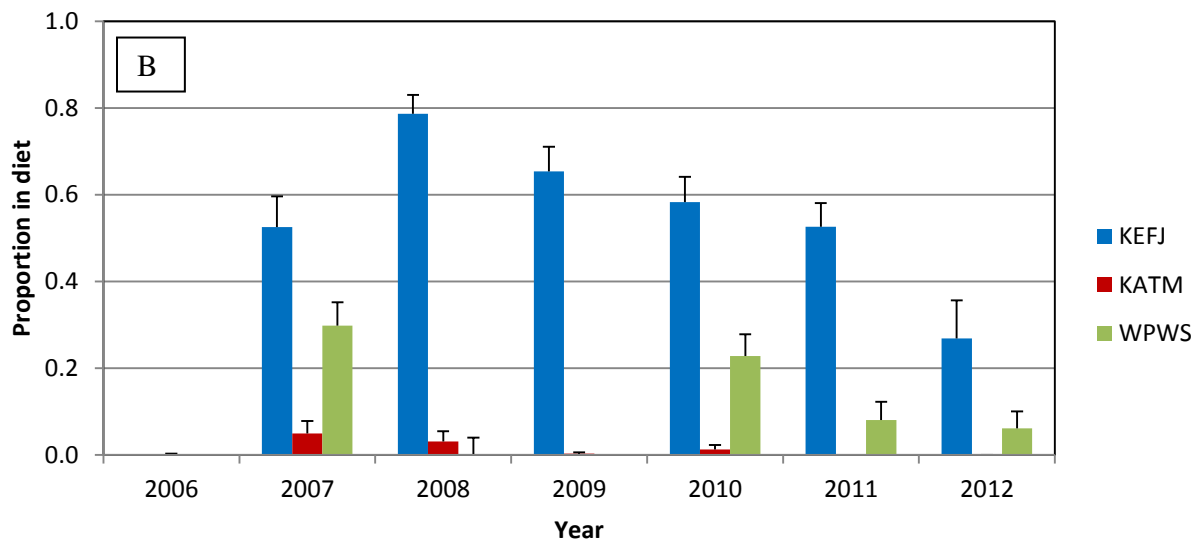
Prey composition of successful sea otter foraging dives in KEFJ, non-clam/mussel items
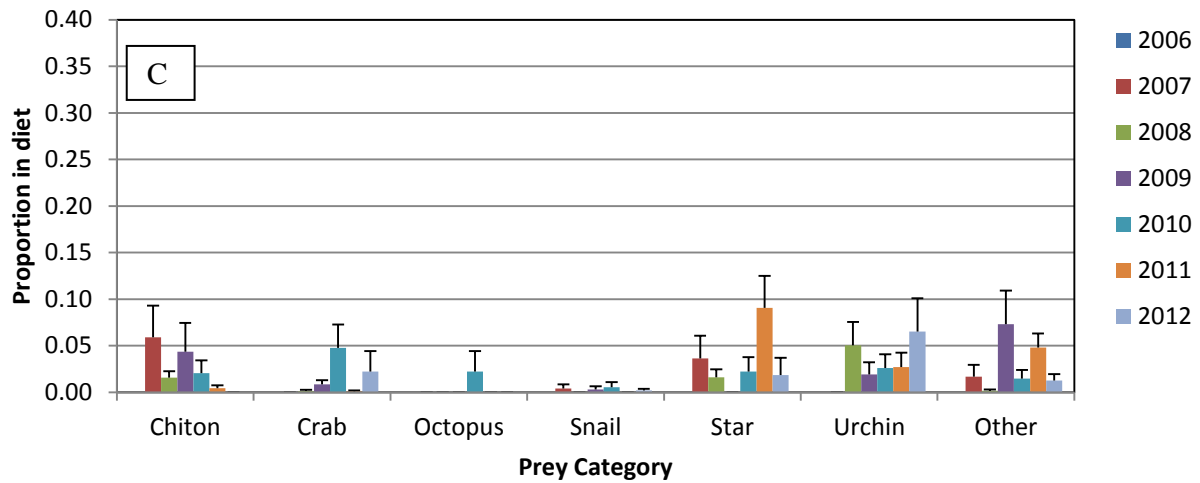
Prey composition of successful sea otter foraging dives in KATM, non-clam/mussel items
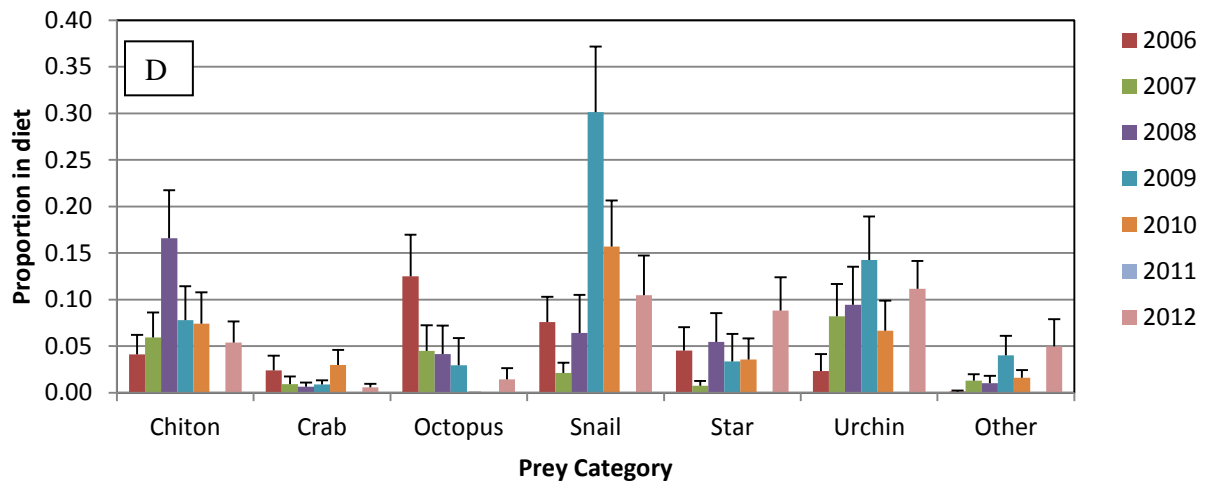
Prey composition of successful sea otter foraging dives in WPWS, non-clam/mussel items
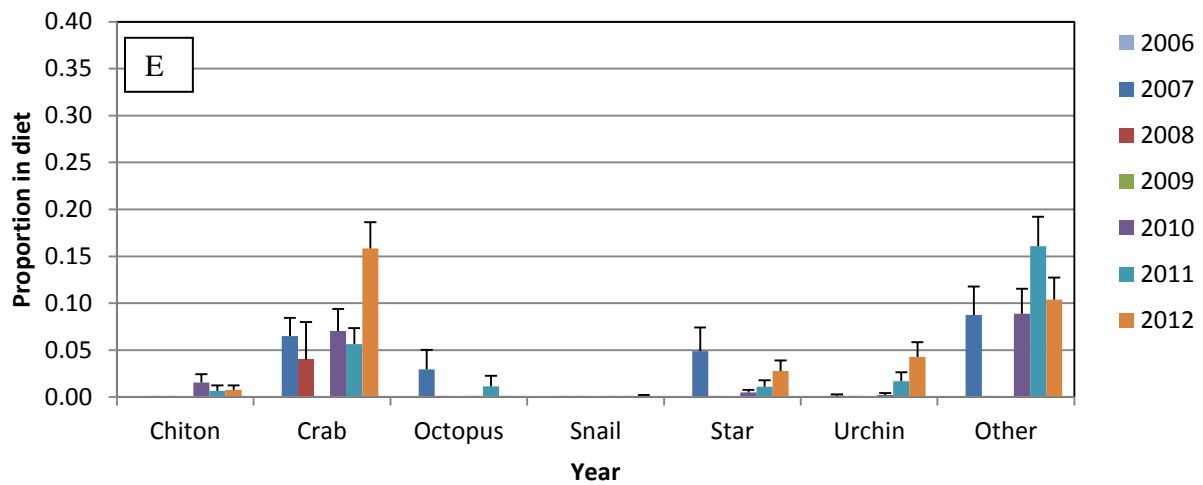
Figure 35. Proportion of identified prey retrieved by foraging sea otters in KEFJ, KATM, and WPWS from 2006 through 2012. Unidentified prey items are not included in these calculations. The "Other" category includes items such as worms, fish, egg cases and other infrequently consumed prey. Additionally, a prey item is only counted towards the proportion once, even if more than 1 of the same item was retrieved on the same dive. Error bars represent one standard error. A. Proportion of prey identified as clam; B. Proportion of prey identified as mussel, primarily *Mytilus trossulus*; C., D., E. Proportions of remaining prey categories for KEFJ, KATM, and WPWS, respectively.

Mean prey size, all years combined
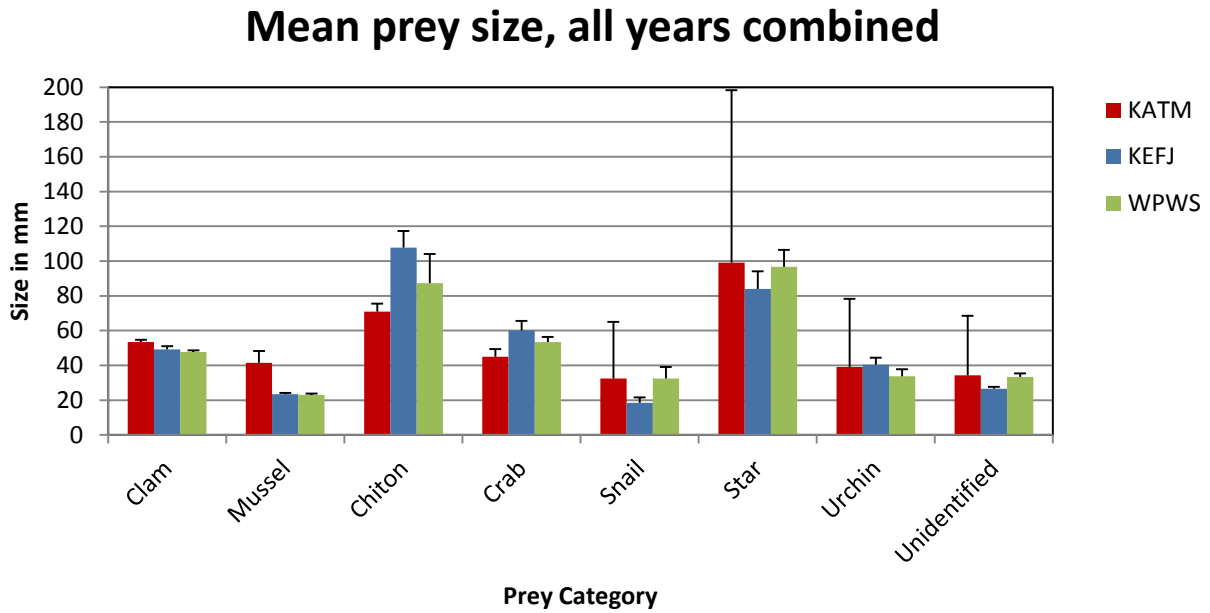
Figure 36. Mean size of prey items recovered by prey category by foraging sea otters in KATM, KEFJ, and WPWS (2006-2012) by region. Sizes from all prey items retrieved were used in the calculations. Error bars represent one standard error.

A | Mean size of clams in sea otter diets

Mean size of mussels in sea otter diets
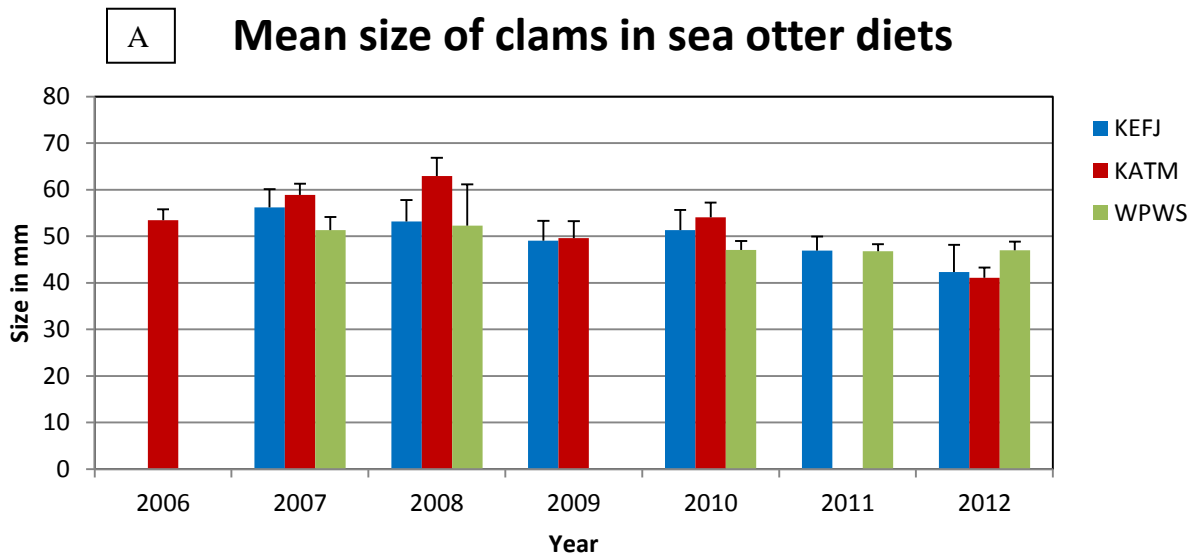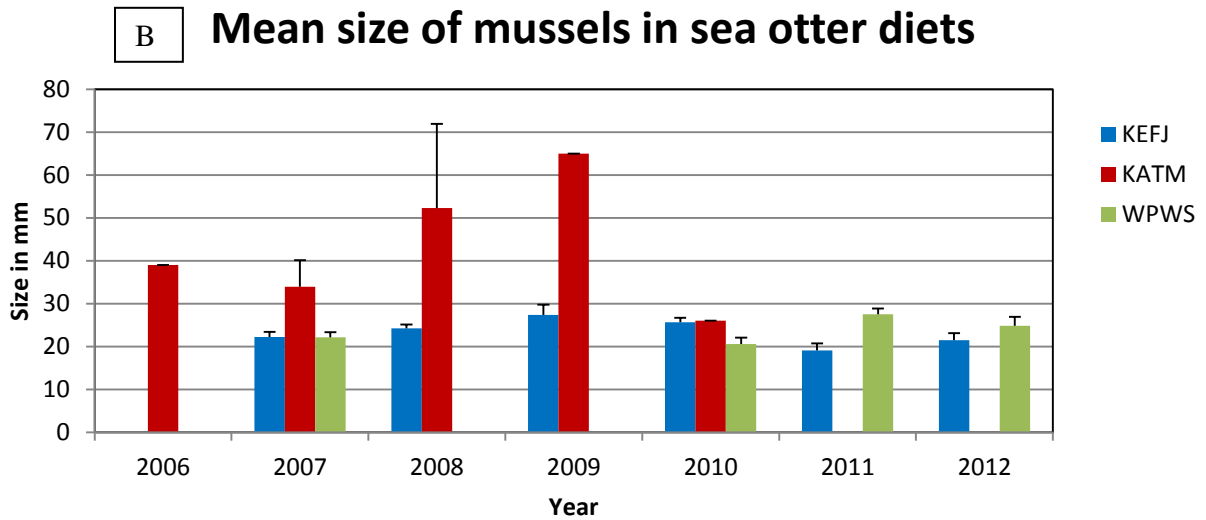
Figure 37. Mean size of A. clams and B. mussels recovered by foraging sea otters in KATM, KEFJ, and WPWS (2006-2012) by year. No mussels were observed being consumed in KATM in 2011 or 2012. Error bars represent one standard error.

Carcass

Since 2006, 179 sea otter carcasses have been successfully aged from KATM and 185 from WPWS (Table 6). The proportions of carcasses in each age all collection years combined are shown in Figure 37. We have also grouped counts into broader age categories to look for changes in proportions. One to two year olds are grouped, representing young, pre-reproductive otters. Three to eight year olds are considered the adult, prime reproductive age otters. Otters older than eight are grouped as aged adults, although still able to reproduce they are likely not as vigorous as prime age otters. The proportions of carcasses in each age category for each year of collection are shown in Figures 38 (KATM) and 39 (WPWS).

Table 3. Summary of sea otter carcasses found in KATM, KEFJ, and WPWS, 2006 - 2012. NS indicates that no surveys for carcasses were conducted.

Year	Number of carcasses		
	KATM	KEFJ	WPWS
2006	36	NS	17
2007	43	0	16
2008	30	0	35
2009	21	0	NS
2010	27	0	18
2011	NS	3	42
2012	22	1	57
All Years	179	4	185

Figure 38. Proportion of sea otter carcasses in each age based on cementum layer analysis.

Figure 39. Proportion of sea otter carcasses collected from KATM in each age category based on cementum layer analysis. KATM was not sampled in 2011.

Tooth ages of sea otter carcasses
from WPWS, 2006-2012
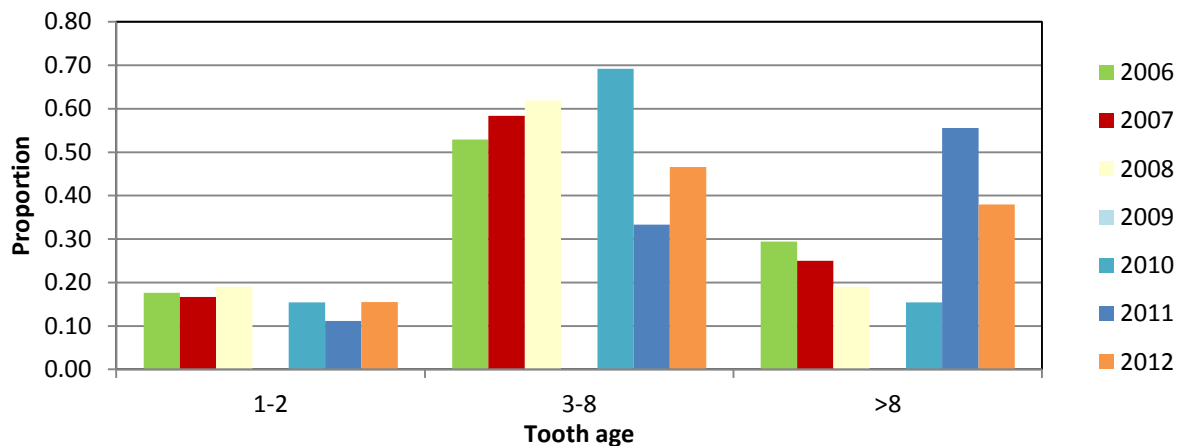
Figure 40. Proportion of sea otter carcasses collected from WPWS in each age category based on cementum layer analysis. WPWS was not sampled in 2009.

Discussion
Aerial survey

This is the second systematic survey of the KATM sea otter population designed to estimate population size. Within four years, the estimated sea otter population has increased by 22% to 8,644 (se=1,243) within the survey boundaries of KATM and an increase in density of > 1 sea otter / km^2 (5.96/ km^2) (Table 4). During the2008 survey, 98% of all sea otters were observed on high density transects and 821 sea otters in 13 large groups at KATM. The estimated detection probability along transects was 0.81 resulting in a correction factor of 1.24 and a total estimated population size of 7,095 sea otters (se = 922). The density of sea otters at KATM across all habitats sampled was 4.89 km^2 in 2008.

To complement that comparison, there are unpublished reports of counts of sea otters along the KATM coast that provide some historic perspective on the process of recovery of sea otters following their extirpation from most of their range during the commercial fur harvest period that ended in 1911 (Kenyon 1969). The first report of sea otters along the KATM coast is from an aerial survey in 1965 when 37 individuals were observed between Kinak Bay and Cape Douglas (Kenyon 1965). Subsequent reports include maximum counts of 443 in June of 1971 near Shakun Is. south of Douglas reef (Prasil 1971), and Goatcher (1994) reported 400-600 sea otters along the KATM coast in 1989. The origin of the initial recolonization of the KATM coast by sea otters is unknown, but most probably resulted from the Kodiak Archipelago, the nearest population known to have survived the commercial fur harvest period.

The current sea otter population that inhabits nearshore waters of KATM occurs at a high density of nearly 6 individuals per km^2. This is substantially higher than the approximate density of 1 individual per km^2 observed elsewhere in the Gulf of Alaska (Bodkin et al. 2008, Bodkin and Udevitz 1999) where populations are thought to be near equilibrium densities. The high density and number of large groups ($\geq$20) encountered both on and off transect (17) with an average size of 48 and a maximum of 150 individuals is consistent with a population increasing in abundance and possibly above long-term equilibrium density.

The KATM sea otter population occurs within the geographic bounds of the Southwest Alaska stock of sea otters (Gorbics and Bodkin 2001) that extends from Cook Inlet to Attu Island in the Western Aleutians. In 2005 this stock was listed as Threatened under the Endangered Species Act (FWS 2005), largely as a result of declines observed in the Aleutian Archipelago and both north and south of the Alaska Peninsula (Doroff et al. 2003). The high density of sea otters we found at KATM strongly suggests that this region currently lies outside the area of decline and that the eastern extent of the decline lies west of KATM (Estes et al. 2010).

Table 4. Sea otter population abundance estimates for KATM, 2008 and 2012. Uncorrected population size is the population size before the correction factor is applied to calculate adjusted population size.

Year	Stratum	Uncorrected Population Size	Correction Factor	Adjusted Population Size	SE	Corrected Density #/km2
2012	High Density	3631	1.88	6808	1053	6.91
	Low Density	256	1.88	479	197	1.03
	Total			**8644[1]**	**1243**	**5.96**
2008	High Density	1120	1.24	4316	399	4.38
	Low Density	23	1.24	225	97	0.48
	Total			**7095[1]**	**922**	**4.89**

[1] Total adjusted population size does not equal the sum of the high and low density strata because the adjusted population size also includes complete counts. Complete counts are treated as a separate stratum in the analysis.

Foraging

Using the methods briefly described above, we were able to estimate sea otter foraging success, prey composition, and mean prey size. The predominant prey retrieved in KATM and WPWS was clams in 2012 and across prior years of data collection. In KEFJ mussels (*Mytilus trossulus*) were the predominant prey item from 2007-2011; however, in 2012 it was clams. These results are based on counts within the diet so we anticipate that further development of the model to analyze rates of energy recovery will allow us to detect ecologically meaningful levels of change based on biomass. Mean size of clams was similar in KEFJ and WPWS and slightly larger in KATM during some years. Mean size of mussels is also similar in KEFJ and WPWS and larger in KATM (2006-2009). However, it may simply indicate that otters in KATM were also feeding on a larger species of mussel, *Modiolus modiolus*. Unidentified prey items were similar in size in KATM and WPWS and smaller in KEFJ. For most other prey items there were too few observations to draw meaningful conclusions. Overall a wide range of prey items was observed across regions. Sea otters display individual preferences in prey selection that can be attributed to prey availability, maternally derived learning and likely several other factors. Since this monitoring protocol has no plans for marking and following individual sea otters' dietary preferences, our analyses will focus on population-level metrics that can be compared over time and to other populations. Unidentified prey is a large component of the diet in all regions. Our developing forage model addresses the unidentified prey component by resampling the known items weighting for other known metrics such as retrieval time, consumption time, and size.

Carcass

Searches for sea otter carcasses continue in KEFJ, KATM, and WPWS. To date, we have not recovered sufficient carcasses from KEFJ to employ age-specific mortality analyses. Discussions are underway to determine ways to improve our carcass recoveries in KEFJ such as adding areas of shoreline to search or searching more frequently to recover carcasses prior to removal by scavengers. Staff in KEFJ and KATM has been searching for carcasses although there are still too few recovered from KEFJ on an annual basis for descriptive analysis. Based on data where years are lumped, WPWS appears to have a higher proportion of older prime age and aged otters dying than KATM. KATM appears to have a higher proportion of prime age adults dying. When examining the annual summaries, lumped into young (age 1-2), adult (age 3-8), and aged (age >8) categories, there are possible trends that warrant more in-depth analyses. Age-0 animals were removed from this analysis under the assumption that age-0 animals are not independent of their mother. In WPWS the proportion of young otters dying appears to be steady over time. The adult category was increasing in proportion through 2008 then declining though 2012. The aged category proportion decreased through 2008 then increased through 2012. In KATM, there was a relatively high proportion of young otters dying in 2006, then that category was steady to declining slightly. The adult category was high in 2009, but stable all other years. The aged category was low in 2006, high in 2010 and 2011 and steady other years. It appears that the differences in proportions are fluctuating between the young and aged categories in KATM, while in WPWS there was a shift between adult and aged groups.

Recommendations

Based on these results, we recommend continued timely aerial surveys of sea otters in the three regions. We also recommend collection of sea otter foraging data with an emphasis on completing the analysis model. Additionally, 50 bouts should be set as the minimum target. Results should be viewed both longitudinally and within the larger framework of known otter foraging studies for context. Sea otter carcass collections should also be continued and the expansion of collection efforts should be seriously considered for KEFJ and KATM. It will be important to build an analysis model that facilitates the inclusion of additional data over time to recognize emerging trends.

Literature Cited

Alaska Shorebird Working Group. 2000. Alaska shorebird conservation plan. Unpublished
Report, U. S. Fish and Wildlife Service, Anchorage, Alaska. 47 pp.

Andres, B. and G. A. Falxa. 1995. Black Oystercatcher (*Haematopus bachmani*) *in* A. Poole and
F. Gill editors. The Birds of North America, No. 155. The Academy of Natural Sciences,
Philadelphia, and The American Ornithologists' Union, Washington, D.C., Philadelphia, PA.

Andres, B. A. 1998. Shoreline habitat use of Black Oystercatchers breeding in Prince William
Sound, Alaska. Journal of Field Ornithology 69: 626-634.

Andres, B. A. 1999. Effects of persistent shoreline oil on breeding success and chick growth in
black oystercatchers. The Auk 116: 640-650.

Barry, J. P., C. H. Baxter, R. D. Sagarin, and S. E. Gilman. 1995. Climate-related, long-term
faunal changes in a California rocky intertidal community. Science 267(5198): 672-675.

Bennett, A.J., W.L. Thompson, and D.C. Mortenson. 2006. Vital signs monitoring plan, Southwest
Alaska Network. National Park Service, Anchorage, AK.

Bodkin, J.L., Ames J.A., Jameson R.J., Johnson A.M., Matson G.E. 1997. Estimating age of sea
otters with cementum layers in the first premolar. J Wildl Manage 61(3): 967–973.

Bodkin, J. L., and M. S. Udevitz. 1999. An aerial survey method to estimate sea otter abundance. G.
W. Garner, S. C. Amstrup, J. L. Laake, B. F. J. Manly, L. L. McDonald, and D. G. Robertson.
Marine Mammal Survey and Assessment Methods. A.A. Balkema, Rotterdam.

Bodkin, J.L., B. Ballachey, T.A. Dean, F.K. Fukuyama, S.C. Jewett, L.L. McDonald, D.H.
Monson, C.E. O'Clair, and G.R. VanBlaricom. 2002. Sea otter population status and the
process of recovery following the 1989 Exxon Valdez oil spill. Marine Ecology Progress
Series 241: 237-253.

Bodkin, J. L., G. G. Esslinger, and D. H. Monson. 2004. Foraging depths of sea otters and
implications to coastal marine communities Marine Mammal Science 20(2):305-321.

Bodkin, J.L., D.H. Monson, and G.G. Esslinger. 2007a. Population status and activity budgets
derived from time-depth recorders in a diving mammal. J. Wildlife Management 71(6): 2034-
2044.

Bodkin, J. L., T. A. Dean, and H. A. Coletti. 2007b. Nearshore Marine Monitoring in the
Southwest Alaska Network of National Parks. National Park Service. Anchorage, AK. 102 pg.

Bodkin, J.L., B.E. Ballachey, G.G. Esslinger, K.A. Kloecker, D.H, Monson, and H.A. Coletti.
2007c. Perspectives on an invading predator—Sea otters in Glacier Bay, *in* Piatt, J.F., and
Gende, S.M., eds., Proceedings of the Fourth Glacier Bay Science Symposium, October 26–
28, 2004: U.S. Geological Survey Scientific Investigations Report 2007-5047, p. 133-136.

Bodkin, J.L., T.A. Dean, H.A. Coletti, and K.A. Kloecker. 2008. Nearshore marine vital signs

monitoring in the Southwest Alaska network of National Parks. Annual report to the US National Park Service, Anchorage, AK. 176 pp.

Bodkin, J.L. 2001a. SOP for conducting marine bird and mammal surveys - Version 4.1: Southwest Alaska Inventory and Monitoring Network. Natural Resource Report NPS/SWAN/NRR—2011/392. National Park Service, Fort Collins, Colorado.

Bodkin, J.L. 2011b. SOP for monitoring black oystercatchers - Version 1.1: Southwest Alaska Inventory and Monitoring Network. Natural Resource Report NPS/SWAN/NRR—2011/391. National Park Service, Fort Collins, Colorado.

Bostrom, C., E.L. Jackson and C.A. Simenstad. 2006. Seagrass landscapes and their effects on associated fauna: a review. Estuar. Coast. Shelf S. 68, 383–403.

Brown, S., C. Hickey, B. Harrington, and R. Gill. 2001. The U.S. Shorebird Conservation Plan, 2nd ed. Brown, S., Hickey, C., Harrington, B., Gill, R. eds. Brown Manomet Center for Conservation Sciences, Manomet, MA.

Burn, D.M. and A.M. Doroff. 2005. Decline in sea otter (*Enhydra lutris*) populations along the Alaska Peninsula, 1986-2001. Fishery Bulletin 103: 270-279.

Calkins, D.G. 1978. Feeding behavior and major prey species of the sea otter *Enhydra lutris* in Montague Straits, Prince William Sound, Alaska. Fishery Bulletin 76: 125–131

Coletti, H.A., J.L. Bodkin, T.A. Dean, and K.A. Kloecker. 2009. Nearshore marine vital signs monitoring in the Southwest Alaska Network of National Parks. Natural Resource Technical Report NPS/SWAN/NRTR—2009/252. National Park Service, Fort Collins, Colorado.

Coletti, H. A., J. L. Bodkin, and G. G. Esslinger. 2011a. Sea otter abundance in Kenai Fjords national Park: results from the 2010 aerial survey: Southwest Alaska Inventory and Monitoring. Natural Resource Technical Report NPS/SWAN/NRTR—2011/417. National Park Service, Fort Collins, Colorado.

Coletti, H. A., J. L. Bodkin, T. A. Dean, and K. A. Kloecker. 2011b. Nearshore marine vital signs monitoring in the Southwest Alaska Network of National Parks: 2010. Natural Resource Technical Report NPS/SWAN/NRTR—2011/497. National Park Service, Fort Collins, Colorado.

Dayton, P. K. 1971. Competition, disturbance and community organization: the provision and subsequent utilization of space in a rocky intertidal community. Ecological Monographs 41: 351-89.

Dean, T.A. M.S. Stekoll, S.C. Jewett, R.O. Smith, and J.E. Hose. 1998. Eelgrass (Zostera marina L.) in Prince William Sound, Alaska: Effects of the Exxon Valdez oil spill. Marine Pollution Bulletin 36: 201-210

Dean T.A., L. Haldorson, D.R. Laur, S.C. Jewett, and A. Blanchard. 2000. The distribution of nearshore fishes in kelp and eelgrass communities in Prince William Sound, Alaska: associations with vegetation and physical habitat characteristics. Environmental Biology of Fishes 57: 271-287.

Dean, T.A., J.L. Bodkin, A.K. Fukuyama, S.C. Jewett, D.H. Monson, C.E. O'Clair, and G.R. VanBlaricom. 2002. Sea Otter (*Enhydra lutris)* Perspective: Mechanisms of Impact and Potential Recovery of Nearshore Vertebrate Predators following the 1989 Exxon Valdez Oil Spill: Part B. Food Limitation and the Recovery of Sea Otters in Prince William Sound. Mar. Ecol. Prog. Ser. 241: 255-270.

Dean, T.A. and J.L. Bodkin. 2011a. Protocol narrative for marine nearshore ecosystem monitoring in the Southwest Alaska Network of National Parks. Natural Resource Report NPS/SWAN/NRR— 2011/449. National Park Service, Fort Collins, Colorado.

Dean, T.A. and J.L. Bodkin. 2011b. SOP for sampling of intertidal invertebrates and algae on sheltered rocky shores - Version 4.6: Southwest Alaska Inventory and Monitoring Network. Natural Resource Report NPS/SWAN/NRR—2011/397. National Park Service, Fort Collins, Colorado.

Dean, T. A., J. L. Bodkin, and H.A. Coletti. 2014. Protocol narrative for marine nearshore ecosystem monitoring in the Gulf of Alaska: Version 1.1. Natural Resource Report NPS/SWAN/NRR— 2014/756. National Park Service, Fort Collins, Colorado.

Doroff, A.M., J.A. Estes, M.T. Tinker, D.M. Burn, and T.J. Evans. 2003. Sea otter population declines in the Aleutian archipelago. Journal of Mammalogy 84(1): 55-64.

Draulans, D. 1982. Foraging and Size Selection of Mussels by the Tufted Duck, *Aythya fuligula*. Journal of Animal Ecology 51(3): 943-956.

Duarte, C.M., 2002. The future of seagrass meadows. Environ. Conserv. 29, 192–206.

Esler, D., J.A. Schmutz, R.L. Jarvis and D.M. Mulcahy. 2000a. Winter survival of adult female harlequin ducks in relation to history of contamination by the Exxon Valdez oil spill. Journal of Wildlife Management 64(3): 839-847.

Esler D., T.D. Bowman, C.E. O'Clair, T.A. Dean, and L.L. McDonald. 2000b. Densities of Barrow's Goldeneyes during winter in Prince William Sound, Alaska, in relation to habitat, food, and history of oil contamination. Water Birds 23(4): 23-429

Esler, D., T. D. Bowman, K. A. Trust, B. E. Ballachey, T. A. Dean, S. C. Jewett, and C. O. O'Clair. 2002. Harlequin Duck Population Recovery Following the Exxon Valdez Oil Spill: Progress, Process, and Constraints. Marine Ecology Progress Series 241: 271-286.

Estes, J.A, R.J. Jameson and E.B. Rhode. 1982. Activity and Prey Election in the Sea Otter: Influence of Population Status on Community Structure. The American Naturalist, Vol. 120, No. 2, pp. 242-258.

Estes, J. A. and D. O. Duggins. 1995. Sea otters and kelp forests in Alaska: generality and variation in a community ecology paradigm. Ecological Monographs 65: 75-100.

Estes, J.A., B.B. Hatfield, K. Ralls, and J. Ames. 2003a. Causes of mortality in California sea otters during periods of population growth and decline. Marine Mammal Science 19(1): 198-216.

Estes, J.A., M.T. Tinker, A.M. Doroff, and D.M. Burn. 2005. Continuing sea otter population declines in the Aleutian archipelago. Marine Mammal Science. 21: 169-172.

Estes, J.A., J.L. Bodkin, and M.T. Tinker. 2010. Threatened southwest Alaska sea otter stock: delineating the causes and constraints to recovery of a keystone predator in the North Pacific Ocean. NPRB Project 717 Final Report.

Faxla, G.A., 1992. Prey choice and habitat use by foraging black oystercatchers. Unpublished Ph.D. dissertation, University of California, Davis.

Frank, P.W. 1982. Effects of Winter Feeding on Limpets by Black Oystercatchers, *Haematopus bachmani*. Ecology 63(5): 1352-1362.

Gelatt, T.S., D.B. Siniff and J.A. Estes. 2002. Activity patterns and time budgets of the declining sea otter population at Amchitka Island, Alaska. Journal of Wildlife Management 66(1): 29-39.

Gerber, L.H., M.T.Tinker, D.F. Doak, J.A. Estes, and D.A. Jessup. 2004. Mortality sensitivity in life-stage simulation analysis: a case study of Southern sea otters. Ecological Applications 14(5): 1554-1565.

Gerrodette, T. 1993. Trends: software for a power analysis of linear regression. Wildlife Society Bulletin 21: 515-516.

Goatcher, B. 1994. Sea otter (*Enhydra lutris*) Information Consolidation Report. Katmai National Park and Preserve. Unpub. Report. King Salmon, AK. 8pp.

Gorbics, C and J.L. Bodkin. 2001. Stock Identity of sea otters in Alaska. Marine Mammal Science 17(3):632-647.

Groves, S. 1984. Chick growth, sibling rivalry, and chick production in American black oystercatchers. The Auk 101: 525-531.

Hahn, T. and M. Denny. 1989. Tenacity-mediated selective predation by oystercatchers on intertidal limpets and its role in maintaining habitat partitioning by *Collisella scabra* and *Lottia digitalis*. Marine Ecology Progress Series. Oldendorf 53(1): 1-10.

Hamilton, D., and T. Nudds. 2003. Effects of predation by common eiders (*Somateria mollissima*) in an intertidal rockweed bed relative to an adjacent mussel bed. Marine Biology 142(1): 1-12.

Hartwick, E. B. 1976. Foraging of the black oystercatcher (*Haematopus bachmani* Audubon). Canadian Journal of Zoology 54(2): 142-155.

Hauxwell, J., J. Cerbian, and I Valiela. 2003. Eelgrass *Zostera mari*na loss in temperate

estuaries: Relationship to land derived nitrogen loads and effect of light limitation by algae. Mar. Ecol. Prog. Ser. 247: 59-73.

Hazlitt, S. and R. W. Butler. 2001. Site fidelity and reproductive success of black oystercatchers in British Columbia. Waterbirds 24(2): 203-207.

Irons, D. B., D. R. Nysewander and J. L. Trapp. 1988. Prince William Sound Sea Otter Distribution in Respect to Population Growth and Habitat Type. Prepared by Alaska Investigations Field Office (Branch of Wetlands and Marine Ecology) and Wildlife Assistance (Marine Bird Project).

Irons, D.B., S.J. Kendall, W.P. Erickson, L.L. McDonald, and B.K. Lance. 2000. Nine years of Exxon Valdez oil spill. Effects of marine birds in Prince William Sound. Condor: 120: 723-737.

Jamieson, G. S., E. D. Grosholz, D. A. Armstrong, and R. W. Elner. 1998. Potential Ecological Implications from the Introduction of the European Green Crab, *Carcinus maenas* (Linneaus), to British Columbia, Canada, and Washington, USA. Journal of Natural History 32(10-11): 1587-1598.

Jewett, S.C., T.A. Dean, R.O. Smith, and A. Blanchard. 1999. The *Exxon Valdez* oil spill: Impacts and recovery in the soft-bottom benthic community in and adjacent to eelgrass beds. Mar Ecol Prog Ser 185: 59-83.

Keough, M.J., G.P. Quinn and A. King. 1993. Correlations between human collection and intertidal mollusk populations on rocky shores. Conservation Biology 7: 378-390.

Keough, M.J. and G.P. Quinn. 1998. Effects of periodic disturbances from trampling on rocky intertidal algal beds. Ecological Applications 8: 141-161.

Kenyon, K.W. 1965. Aerial surveys of sea otters and other marine mammals, Alaska Peninsula and Aleutian Islands, 19 April to 9 May 1965. U.S. Fish and Wildlife Service, Seattle, WA.

Kenyon, K.W. 1969. N. American Fauna 68: The Sea Otter in the Eastern Pacific Ocean. US Fish and Wildlife Service. pp. 352.

Kingsford, M.J., A.J. Underwood and S.J. Kennelly. 1991. Humans as predators on rocky reefs in New South Wales, Australia. Marine Ecology Progress Series72: 1-14.

Knox, G.A. 2000. The Ecology of Seashores. CRC Press LLC, Boca Raton, FL. 557 pp.

Larkum, W.D., Orth, R.J., Duarte, C.M., 2006. Seagrasses: Biology Ecology and Conservation. Springer, The Netherlands.

Lentfer, H.P. and A.J. Maier. 1995. Breeding ecology of the black oystercatcher in the Beardslee Island region of Glacier Bay National Park. In Proceedings of the Third Glacier Bay Science Symposium Proceedings of the Third Glacier Bay Science Symposium.

Lewis, J. 1996. Coastal benthos and global warming: strategies and problems. Marine Pollution Bulletin 32: 698-700.

Lewis, T.L., D. Esler, and W.S. Boyd. 2007. Effects of predation by sea ducks on clam abundance in soft-sediment intertidal habitats. Marine Ecology Progress Series 329: 131-144.

Liddle, M. J. 1975. A selective review of the ecological effects of human trampling on natural ecosystems. Biological Conservation 7: 17-36.

Lindberg, D.R., K.I. Warheit and J.A. Estes. 1987. Prey Preference and Seasonal Predation by Oystercatchers on Limpets at San Nicolas Island, California USA. Marine Ecology. 39: 105-113.

Lindberg, D.R., J.A. Estes and K.I. Warheit. 1998. Human influences on trophic cascades along rocky shores. Ecological Applications 8: 880-890.

Marsh, C.P. 1986a. The Impact of Avian Predators on high intertidal limpet populations. J. Exp. Mar. Biol. Ecol. 104: 185-201.

Marsh, C.P. 1986b. Rocky Intertidal Community Organization: The Impact of Avian Predators on Mussel Recruitment. Ecology 67: 771-786.

McConnaughey, T. & McRoy, C.P. 1979. C13 label identifies eelgrass (Zostera marina L.) carbon in an Alaskan estuarine food web. Mar. Biol. 53:263-269.

McRoy, C.P. 1968. The distribution and biography of Zostera marina L. (eelgrass) in Alaska. Pac. Sci. 22: 507-513.

McRoy, C.P. 1970. Standing stocks and other features of eelgrass (Zostera marina L.) populations on the coast of Alaska. J. Fish. Res. Bd. Can. 27, 1811-1821.

Meese, R.J. 1990. Effects of Predation by Birds on Rocky Intertidal Community Structure. Bull. Ecol. Soc. Am. 71: 250.

Meire, P.M. 1993. The Impact of Bird Predation on Marine and Estuarine Bivalve Populations: A selective Review of Patterns and Underlying Causes Bivalve Filter Feeders in Estuary. In Bivalve Filter Feeders in Estuarine and Coastal Ecosystem Processes. Vol. G33 Heidelberg, Germany: Springer Verlag.

Menge, B.A. and G.M. Branch. 2001. Rocky intertidal communities. in Marine Community Ecology. Bertness, Brown University, University of California at Santa Barbara, Georgia Institute of Technology. Marine Community Ecology. Sinauer Associates, Inc., Sunderland, Massachusetts.

Monson, D. H., D. F. Doak, B. E. Ballachey, A. M. Johnson, and J. L. Bodkin. 2000. Long-term impacts of the Exxon Valdez oil spill on sea otters assessed through age-dependent mortality patterns. Proceedings of the National Academy of Sciences 97: 6562-6567.

Neckles, H.A., Short, F.T., Barker, S., Kopp, B.S., 2005. Disturbance of eelgrass *Zostera marina* by commercial mussel *Mytilus edulis* harvesting in Maine: dragging impacts and habitat recovery. Mar. Ecol.-Prog. Ser. 285, 57–73.

O'Clair, C.E. and S. D. Rice. 1985. Depression of feeding and growth rates of the seastar

Evasterias troschelii during long-term exposure to the water-soluble fraction of crude oil. Marine Biology 84: 331-340.

O'Clair, R.M and C.E. O'Clair. 1988. Southeast Alaska Rocky Shores: Animals. Plant Press, Auke Bay, Alaska.

Olesen, B., 1996. Regulation of light attenuation and eelgrass *Zostera marina* depth distribution in a Danish embayment. Mar. Ecol.Prog. Ser. 134, 187–194.

Peterson, C.H. 2001. The "Exxon Valdez" oil spill in Alaska: acute, indirect and chronic effects on the ecosystem. Advances in Marine Biology 39: 1-103.

Peterson, C H., S.D. Rice, J.W. Short, D. Esler, J.L. Bodkin, B. E. Ballachey, and D.B. Irons. 2003. Long-term ecosystem response to the Exxon Valdez oil spill. Science 302: 2082-2086.

Povey, A. and M.J. Keough. 1991. Effects of trampling on plant and animal populations on rocky shores. Oikos 61: 355-368.

Power, M.E., D. Tilman, J.A. Estes, B.A. Menge, W.J. Bond, L.S. Mills, G. Daily, J.C. Castilla, J. Lubchenco, and R. T. Paine. 1996. Challenges in the quest for keystones. BioScience 46: 609-620.

Prasil, R. 1971. Sea and Coastline Surveys Katmai National Monument, July 1969 through June 26, 1971. National Park Service, Alaska Region Natural Resources Survey and Inventory Report AR-71/02. National Park Service, Anchorage, AK. 9pp.

Riedman, M.L. and J.A. Estes. 1990. The Sea Otter (*Enhydra lutris*): Behavior, Ecology, and Natural History. Washington, D.C.: US Fish and Wildlife Biologocal Report 90. 126p.

Sagarin R.D., J.P. Barry, S.E. Gilman, and C.H. Baxter. 1999. Climate related changes in an intertidal community over short and long time scales. Ecological Monographs 69: 465-490.

Schiel, D.R., and D.I. Taylor. 1999. Effects of trampling on a rocky intertidal algal assemblage in southern New Zealand. J. Exp. Mar. Biol. & Ecol. 235: 213-235

Schultz, S.T. 2008. Seagrass monitoring by underwater videography: Disturbance regimes, sampling design, and statistical power. Aquat. Bot. 88: 228-238.

Short, F. T., and S. Wyllie-Echeverria. 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation 23: 17-27.

Short, F.T., Koch, E.W., Creed, J.C., Magalhaes, K.M., Fernandez, E., and Gaeckle, J.L., 2006. Seagrass net monitoring across the Americas: case studies of seagrass decline. Mar. Ecol. 27, 277–289.

Sousa, W. P. 1979. Experimental Investigations of Disturbance and Ecological Succession in a Rocky Intertidal Algal Community. Ecological Monographs. 49(3): 227-254.

Stevens, D.L. and A.R. Olsen. 2004. Spatially Balanced Sampling of Natural Resources. Journal

of the American Statistical Association 99: 262-278.

Terrados, J., M. Grau-Castella, D. Pinol-Santina, P. Riera-Fernandez 2006. Biomass and primary production of a 8–11 m depth meadow versus 3 m depth meadows of the seagrass *Cymodocea nodosa* (Ucria) Ascherson. Aquat. Bot. 84: 324–332.

Thayer, J. K. and R.C. Phillips 1977. Importance of eelgrass beds in Puget Sound. Mar. Fish. Rev. 36: 27-39.

Tinker, M.T., D.F. Doak, J.A. Estes, B.B. Hatfield, M.M. Staedler, and J.L. Bodkin. 2006. Incorporating diverse data and realistic complexity into demographic estimation procedures for sea otters. Ecological Applications 16: 2293-2312.

Tinker, M.T., G. Bentall, and J.A. Estes. 2008. Food limitation leads to behavioral diversification and dietary specialization in sea otters. Proceedings of the National Academy of Sciences 105: 560-565.

Trust, K. A., D. Esler, B. R. Woodin and J. J. Stegeman. 2000. Cytochrome P450 1A induction in sea ducks inhabiting nearshore areas of Prince William Sound, Alaska. Marine Pollution Bulletin 40: 397-403.

Walker, D. L., Lukatelich, R. J., Bastyan, G. & McComb, A. J. 1989. Effect of boat moorings on seagrass beds around Perth, Western Australia. Aquat. Bot. 36, 69-77.

Webster, J. D. 1941. Feeding habitats of the black oyster-catcher. The Condor 43: 175-180.

Wootton, J. T. 1992. Indirect Effects, Prey Susceptibility, and Habitat Selection: Impacts of Birds on Limpets and Algae. Ecology 73(3): 981-991.

VanBlaricom., G.R. 1988. Effects of foraging by sea otters on mussel-dominated intertidal communities. Pages 48-91 *in* G.R. VanBlaricom and J.A. Estes (eds.) The Community Ecology of Sea Otters. Springer Verlag, Berlin, Germany.

VanBlaricom, G.R. and J.A. Estes, eds. 1988. The Community Ecology of Sea Otters. New York, NY. Springer-Verlag.

VanBlaricom, G. R. and J. A. Estes. 1988. The Community Ecology of Sea Otters. Springer-Verlag, Berlin. National Park Service. 2010. Instructions to authors — Natural Resource Technical Report, Natural Resource Technical Report, and Natural Resource Data Series: version 3.1. Natural Resource Technical Report. NPS/NPRC/IMD/NRTR—2010/256. National Park Service, Fort Collins, Colorado.

Vermeer, K., K. H. Morgan and G. E. J. Smith. 1992. Black oystercatcher habitat selection, reproductive success, and their relationship with Glaucous-winged gulls. Colonial Waterbirds 15: 14-23.

Natural Resource Stewardship and Science
1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525

www.nature.nps.gov